A review of the ecological role of forage fish and management strategies

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by
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EXECUTIVE SUMMARY

The report reviews the ecological role of forage fish globally and in Canada, and the policy directed at their management, with a focus on the consumption requirements of predators. The report is intended for policy makers, managers and biologists who are engaged in providing advice or making decisions concerning the management of forage species.

Ecologically essential
Forage fishes are short-lived, highly productive, abundant, and are represented by only a few species in an ecosystem. Typically, their abundance varies considerably in time, influenced by environmental conditions and fishing. They play an important role in the ecosystem, being an essential link between planktonic production and higher trophic level predators. These species can be crucial as forage base for dependent predators, in determining the breeding success of marine birds and the condition and even reproductive capacity of mammals and fish predators (e.g. cod). Large decreases in the abundance of these species beyond normal fluctuations risk modifying the foodweb, changing energy pathways, in some cases, towards fewer species and ultimately to an alternate state from which it is difficult to recover.

Economically important
Economically, forage fishes are important for the large directed fisheries they support but more importantly for their major (~20%) contribution to the production of all marine fisheries (large predators).

Reference points and harvest control rules are not the ultimate management system
Reference points are useful but do not constitute the end of the process for forage fish management given that they are defined with large uncertainty and bias, do not generally include the consumption requirements of predators and can change due to environmental variability or climate change. In Canada, some reference points are set in an ad hoc manner (or absent) due to lack of information.
The key points for successful management are: low exploitation rate, simple rules, and implementing the required management actions. In Canada and around the world, numerous examples show that scientific recommendations are overruled in favour of short-term social and economic considerations and doing so, jeopardize stock recovery and dependent fishery communities. There can be little hope that ecosystem-based management will solve the problems of overexploitation in fisheries if we cannot even implement single-species recommendations.

A pragmatic approach towards an ecosystem-based management
Case studies show that current single-species management should be complemented with ecosystem information (e.g. predation and environmental conditions) as required. In most cases, including predation considerations results in a recommended reduction in exploitation rate and quotas. At the moment, most reference points and harvest control rules do not account for the role of forage fish in the ecosystem and the predators' needs. In spite of numerous policy and strategic documents, Canada is part of the group of countries that do not account for the special situation of forage fish in terms of management.

Need for information
Sooner or later, environmental factors and/or predation have to be taken into account for all ecosystems. This became obvious earlier in upwelling areas and extreme environments such as the Antarctic, the Barents Sea, the Baltic, or the Black sea. Historically, the most successful cases were those ecosystems where ecological data were gathered to answer the questions posed by collapses. In Canada, the lack of information on forage fish is pervasive.

SOMMAIRE DE GESTION

Le rapport examine le rôle écologique des espèces de poissons-fourrages au Canada et à l'échelle planétaire, ainsi que les politiques qui régissent leur gestion, et il met l'accent sur les exigences relatives à la consommation des prédateurs. Le rapport s'adresse aux décideurs, aux gestionnaires et aux biologistes qui donnent des conseils ou prennent des décisions en matière de gestion des espèces fourragères.

Impératif sur le plan écologique
Les poissons-fourrages ont une durée de vie courte, sont prolifiques, se trouvent en quantité abondante et ne sont représentés que par un petit nombre d'espèces dans un écosystème. En règle générale, leur abondance varie considérablement dans le temps et elle est influencée par les conditions environnementales et la pêche. Ils jouent un rôle important dans l'écosystème, en ce qu'ils constituent un lien essentiel entre la production planctonique et les prédateurs des échelons trophiques supérieurs. Parce que ces espèces servent de nourriture de base aux prédateurs dépendants, elles peuvent jouer un rôle capital dans le succès de la reproduction d'oiseaux marins, mais aussi sur les conditions des mammifères et des poissons qui se nourrissent d'autres poissons (p. ex. morue), voire sur leur capacité reproductrice. Si la diminution de l'abondance des espèces fourragères est importante au point de descendre sous le niveau des fluctuations normales, le réseau trophique risque d'être modifié, changeant ainsi les flux d'énergie et entraînant, dans certains cas, une réduction du nombre d'espèces et, au final, un état différent duquel un rétablissement est difficile.

Importance sur le plan économique
Les poissons-fourrages sont importants pour les pêches dirigées vers ces espèces, mais plus encore pour leur contribution majeure (environ 20 %) à la production de toutes les pêches marines (grands prédateurs).

Les points de référence et les règles de contrôle des prises ne constituent pas le système de gestion suprême
Les points de référence sont utiles, mais ils ne suffisent pas à déterminer le processus de gestion des poissons-fourrages. En effet, ils sont définis avec d'importantes incertitudes et un certain biais, ils ne sont en général pas établis en tenant compte des exigences en matière de consommation des prédateurs et ils peuvent fluctuer en fonction de la variabilité environnementale ou des changements climatiques. Au Canada, certains points de référence sont établis de façon ponctuelle (ou sont absents) en raison du manque d'information.
Les principaux éléments d'une gestion réussie sont les suivants : faible taux d'exploitation, règles simples et mise en place des mesures de gestion nécessaires. Au Canada et dans le monde, de nombreux exemples montrent que les recommandations scientifiques sont écartées en faveur de

considérations socio-économiques à court terme, ce qui met en péril le rétablissement des stocks et compromet les collectivités qui dépendent de la pêche. On peut difficilement espérer qu'une gestion écosystémique règle les problèmes de la surexploitation si nous ne pouvons pas même mettre en œuvre des recommandations relatives à une seule espèce.

Approche pragmatique vers une gestion écosystémique

Des études de cas montrent que l'approche de gestion actuelle axée sur les espèces uniques devrait être, au besoin, complétée par des données sur les écosystèmes (p. ex. prédation et conditions environnementales). Dans la plupart des cas, quand la prédation est prise en compte, une réduction des taux d'exploitation et des quotas est recommandée. Pour le moment, la plupart des points de référence et des règles de contrôle des prises ne tiennent pas compte du rôle des poissons-fourrages dans l'écosystème et des besoins des prédateurs. Malgré le grand nombre de politiques et de documents stratégiques, le Canada fait partie du groupe de pays qui ne tiennent pas compte de la situation particulière des poissons-fourrages en matière de gestion.

Besoin d'information

Tôt ou tard, les facteurs environnementaux et la prédation devront être pris en compte pour tous les écosystèmes. Cela est devenu évident dans les zones de remontée des eaux ou les environnements extrêmes comme en Antarctique, dans la mer de Barents, la mer Baltique ou la mer Noire. Par le passé, les situations qui ont été les mieux gérées concernaient des écosystèmes pour lesquels on possédait des données écologiques afin de répondre aux questions posées par les effondrements des stocks. Au Canada, le manque de données sur les poissons-fourrages est généralisé.

Introduction

Forage species occur at mid-trophic levels in the food chain and include krill and fish such as sardines, anchovies, herring, capelin, sandeel (sand lance), menhaden, characterized by a small body size, rapid growth, relatively large population size, and mid-trophic level. These species support the largest fisheries in the world, especially in upwelling ecosystems and overfishing is common worldwide (Alder and Pauly 2006; Pinsky et al. 2011). The majority of landings are processed into fish meal and oil used in aquaculture and agriculture, some landings are used for human consumption (Alder and Pauly 2006), and as bait for other fisheries (e.g. lobster in Maine and eastern Canada) (Grabowski et al. 2010). Forage species play a critical role in transferring energy from plankton to large predatory species of fish, mammals and birds (DFO 2010c; Pikitch et al. 2012a) and are represented by only a few species in each ecosystem. Commercial forage fish fisheries are often in competition with the needs of predatory species that depend on forage species for a large part of their nutritional needs.

The report represents the first component of a Fisheries and Oceans Canada (DFO) Strategic Program for Ecosystem Research and Advice (SPERA) project that reviews the ecological role of forage species and the principles of policy directed at their management in Canadian waters where they occur. This document will provide a basis to identify relevant and targeted questions related to ecosystem-based management of forage fish in Canada. Based on published documents and consultation of experts when necessary, the review will:

- Synthesise the latest understandings of the role of forage fish in ecosystems around the world;
- Synthesise the principles of management drawn from published studies including the reports produced by the Lenfest, FACTS, and European Union Working Groups;
- Describe management practices and tools for forage fish in other parts of the world;
- Briefly describe the Canadian situation and management practices on both coasts, concentrating on herring, but including other species such as sand lance, alewife, capelin, mackerel and eulachon as needed;
- Compare Canadian practices and tools with those used in other parts of the world;
- Derive lessons learned and make recommendations for the science advice and management of Canadian Forage fish fisheries.

FORAGE FISH, THEIR ECOLOGICAL ROLE AND THEIR ECOSYSTEM

This report concerns mainly forage species (fish and krill) living on continental shelves and slopes (Figure 1). It does not take into account mesopelagics, the forage fish of the open sea. Mackerel is generally not considered as a forage fish since it is piscivorous as adult. It is included in the case studies in this report because it feeds on plankton on the Atlantic coast, and its juveniles are found in mixed schools with herring (M. Power, DFO, unpublished data). Euphausids (krill) are small shrimp-like crustaceans that can aggregate in dense layers. They are present in most ecosystems and part of the large zooplankton community in most ecosystem models. Krill feed on smaller zooplankton and constitute an important link between plankton and forage fish as well as predators. In Antarctica though, they are the main mid-level trophic species and are the object of a dedicated management strategy, and as such, are worth a section as a management case study.

Several forage fishes are purely pelagic (e.g. sardine, anchovy) while others dwell in sandy bottoms during the day and rise in the water column at night (e.g. sand eel). Others are pelagic but spawn on the bottom (e.g. herring). Thus, these species occur in various types of habitats although they are all largely dependent on the planktonic community for food.

The following sections review the roles of forage fish in the ecosystem starting with the concept of wasp-waist ecosystems, describing the particular role of small pelagic fish (e.g. anchovy, sardine, herring), a group of forage fish that have been shown to have a pivotal importance in these type of food webs (Cury et al. 2000; Bakun 2006).

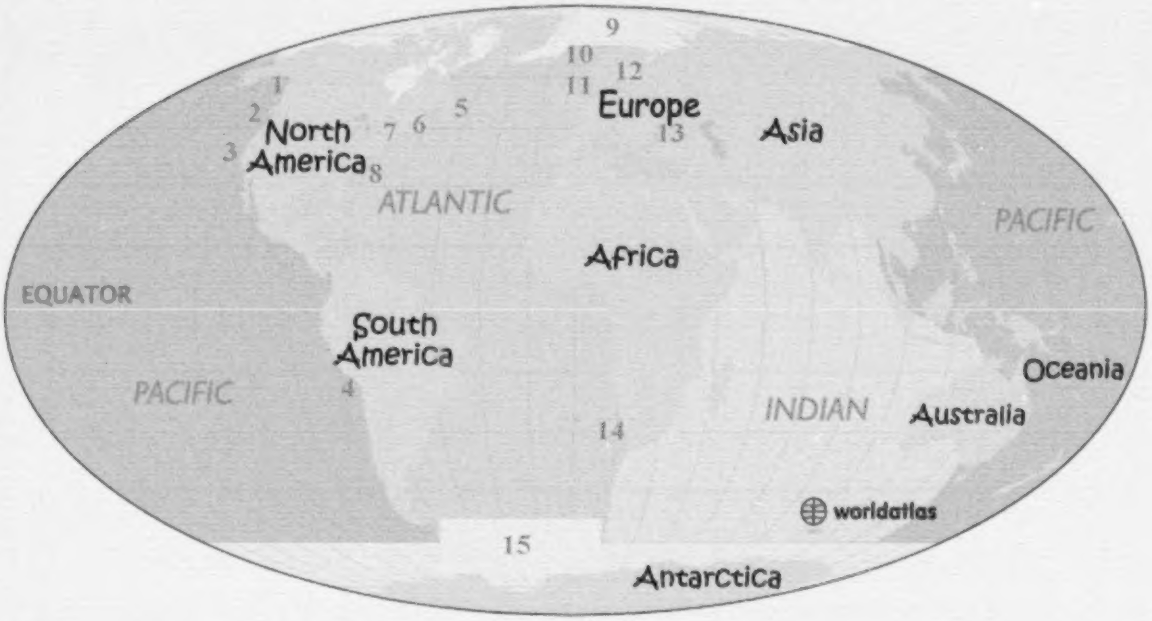

Figure 1. Location of various forage species mentioned in this report.

(map from http://www.worldatlas.com/aatlas/infopage/oceans.htm).
1. Alaskan herring; 2. British Columbia herring; 3. NE Pacific sardine; 4. Peruvian anchovy; 5. Newfoundland capelin; 6. Canadian mackerel and herring stocks; 7. Gulf of Maine herring; 8. menhaden; 9. Barents Sea capelin; 10. Norwegian herring; 11 North Sea species (herring, Norway pout, sandeel, sprat); 12. Baltic Sea; 13. Black Sea; 14. South African sardine and anchovy); 15. Antarctic krill.

WASP-WAIST CONTROL

Forage species are highly productive and short-lived, occur in large abundances, and occur at mid-trophic levels. Small pelagic fish in particular are mobile, able to relocate, their geographic distribution expands or contracts rapidly in response to environmental conditions and/or changes in biomass levels (Bakun et al. 2010). Typically, only one or a few species of small pelagics are present in an ecosystem and they play the same general role of channelling the flow of energy from plankton to higher trophic level predators. The term wasp-waist ecosystem conveys the exceptional concentration of biomass in a few species at the mid-trophic level compared to the species richness in invertebrate plankton and in predators species (Rice 1995; Bakun 1996; Cury et al. 2000). It was then hypothesized that in most ecosystems (Bakun et al. 2010), small pelagics can exert top-down control of zooplankton and a bottom-up control on predators, in a "wasp-waist" structure controlled by very few species from the middle trophic level (Bakun 1996). This typically occurs in coastal, highly productive systems with short food chains forced from the bottom by strong nutrient supply mechanisms (Hunt and McKinnell 2006). In these ecosystems, changes in abundance and switches in dominance are common and impact dependent predators (Chavez et al. 2003; Cury et al. 2004; Bakun et al. 2010; de Moor et al. 2011). This conclusion was based on a few examples in ecosystems showing inverse relationships between small pelagics and zooplankton or small pelagics and dependent predators (Cury et al. 2000).

Bottom-up, top-down and wasp-waist controls are 3 mechanisms of ecosystem functioning that can combine at times without excluding other mechanisms that structure ecosystems. However, the relative strength of top-down and bottom-up control and the strength of the trophic cascades[1] is highly variable (Hunt and McKinnell 2006; Johannesen et al. 2012), dependent on the relative biomass of small pelagics (Bakun 2006), on climate regime (e.g. Litzow and Ciannelli 2007) and the complexity of the ecosystem (e.g., trophic cascades are stronger in simpler ecosystems such as the Baltic Sea or Black Sea, Daskalov 2002; Peck 2011a). Thus the wasp-waist control may be an over-simplified representation of complex ecosystems that ignores other important mechanisms and species.

In the North Sea, on average, trophic control is bottom up in the spring during the phytoplankton bloom and top-down dominated in the summer, but these patterns can change with environmental regimes (a shift was detected in the early 1980s) and interactions between planktivorous and piscivorous fish (Peck 2011a). Also, wasp-waist effects can be compartmented in time, herring being an important prey in the winter (bottom-up) for seabirds, while top-down control was found between herring and krill, and between sprat and one species of copepod (Fauchald et al. 2011).

Zooplankton are highly vulnerable to predation and the indirect effects of fishing (i.e., competition for zooplankton decreases with increased fishing (Cury et al. 2000). However, the abundance of planktonic species is also dependent on environmental conditions of stochastic and cyclic nature which in turn influences the food available to small pelagics (Hunt and McKinnell 2006).

ROLE AS PREY

Some dependent predators, such as marine birds, are negatively impacted by the decreased abundance of forage fishes (Hunt and McKinnell 2006). For instance, the capelin collapse in the Barents Sea led to the abandonment of seabird colonies and the invasion of coastal areas by harp seals in search of food (Gjøsaeter et al. 2012). Other examples of the effect of a decline or change in phenology of key forage fish are documented in several ecosystems encompassing the Southern Benguela upwelling (Crawford et al. 2011), the Antarctic (Reid et al. 2005), the Barents Sea (Hjermann et al. 2004) and more diverse ecosystems such as the North Sea (ICES 2007b; Fauchald et al. 2011) and Newfoundland (Davoren and Montevecchi 2003; Davoren et al. 2012).

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[1] Trophic cascades describe the reaction of lower trophic levels to changes in biomass of predators. In a top-down situation, the impact of top-predators propagates down the food web and ultimately influences primary production (Carpenter et al. 1985). For instance, the removal of large predators that results in the increase of the middle-level prey, which in turn results in the decrease in biomass of plankton species (low trophic level).

Modelling studies show that a lack of forage fish can lead to a displacement of mortality for prey to cannibalism in large predatory fish (ICES 2007b). In the North Sea, the condition of cod and of several large fish predators is linked to the amount of sandeel in their diet (ICES 2010b). In the Barents Sea, cod cannibalism increased during the capelin collapse (Hjermann et al. 2004). Cod weight-at-age was found to be related to the biomass of herring and sprat present in the Baltic (ICES 2007b).

Predators can also control forage fish (top-down control), particularly when they are at intermediate population sizes, sufficiently abundant to attract predators but not so numerous as to swamp the numerical response of the predator (Bakun 2006). In some ecosystems, the depletion of top-predators coincides with an increase in planktivorous fish (e.g. Bundy and Fanning 2005; Frank et al. 2005; Bundy et al. 2009), but not in systems like the North Sea, where the depletion of some of the forage fish species did not lead to trophic cascades, perhaps because of its species rich communities (Peck 2011a). Indeed, the North Sea is home to at least four different forage fishes that occupy different habitats and preferred environmental conditions. Predators are thus not dependent on one are two main species.

ROLE AS PREDATOR

While forage fishes share a general role in marine ecosystems, individual species display a wide array of feeding behaviours. Although sardine and anchovy can both feed by filtering or picking particulates, the thresholds to switch between feeding modes differ (van der Lingen et al. 2006). Anchovy, characterised by large gaps between its gill rakers, will mainly particulate-feed on a large variety of prey, but cannot access phytoplankton. Because of the small gaps between gill rakers, sardine is able to more efficiently remove small particles including phytoplankton by filtering, its main feeding mode (van der Lingen et al. 2006). Thus, although both are zooplanktivorous, their diet overlap is minimised by prey size resource partitioning (van der Lingen et al. 2006) and habitat preference (Bertrand et al. 2004).

The EU Forage Fish Interaction project (FACTS) has completed and reviewed studies on the food web characteristics in European waters and show important differences in behaviour between species. For instance, sprat, an obligate particulate feeder, may have difficulty meeting basic requirements when competition is high due to high density of conspecifics or low biomass of prey and show low body condition as a result (Peck 2011a). Forage fishes often have overlapping diets but may have different habitat requirements and occupy slightly different niches. For instance, in the Baltic Sea, sprat and herring adults were found to depend on copepods, but preferred slightly different species. The frequent ontogenic change in diet between juvenile and adult reduces intra-specific competition, but it does not occur in sprat and thus, they can suffer from density-dependent food limitation. In the North Sea, the six forage fishes considered show considerable diet overlap but the magnitude of their interactions depends on their spatial and habitat distribution (Peck 2011a).

COMPLEX RELATIONSHIPS

The role of forage fish in the ecosystem is more complex than simple predator-prey relationships as they can also prey on competing species (intraguild predation) and even on early life-stages of their predators[2]. Forage fish have been found to mix with other species of similar stage/sizes, especially when their population size is decreasing (Bakun 2006; Zwolinski and Demer 2012). In the North Sea, adult cod eat herring but only adult herring (as opposed to other species) eat cod eggs because it selects large eggs compared to other forage species (Peck 2011a). This can promote different alternative stable states of the ecosystems, which also means that cod or herring recovery (from one stable state to another) may be difficult (Fauchald 2010; Peck 2011a). Juvenile herring are known to have a clear preference for larvae of their own species and in fact, cannibalism may be responsible for increased herring larvae mortality and reduced recruitment after 2000 in the North Sea (Corten 2013).

In the Barents Sea, ecological studies have revealed complex trophodynamic relationships between herring, capelin (the main forage fish) and cod. Capelin collapses are viewed as natural fluctuations caused by consecutive recruitment failures triggered by increased larval predation from large cohorts of herring entering the Barents Sea (Gjøsaeter et al. 2012). After the capelin collapse, cod predation induced a lag in capelin recovery, showing the importance of top-down control in this situation. Both herring and cod recruitment is influenced by climate and has an effect on capelin. Cod depends heavily on capelin before spawning and thus the capelin collapse resulted in an increase in cannibalism and decrease in cod body condition (Hjermann et al. 2004). In contrast, herring feed on larvae but is not dependent on capelin and migrate to the Barents Sea irregularly for reasons not understood (Hjermann et al. 2004). Cod and fishing are competing for capelin, and overharvesting of cod will increase the impact on cod predation (Hjermann et al. 2004). However, the link between cod and capelin has decreased over the last 20 years owing to the warming of the Barents Sea and the increase in abundance of warm water pelagic species and hence diversity (Johannesen et al. 2012).

Herring predation on sandeel larvae is also suspected to have caused the decline in recruitment for this species and in turn, the decline in sandeel has caused decline in breeding success in several seabirds in the Shetlands area (ICES 2007b). These few examples show that trophodynamics directly affect fish production.

EFFECTS OF ENVIRONMENTAL FACTORS

Climate regimes can have direct and indirect effects on ecosystems and forage fish productivity and distribution (e.g. Peck 2011b). Changes in water temperature driven by movements of water masses, for instance, influence the distribution of species and predator-prey relationships. In Iceland since the late 1990s, the amount of inflow of warm water from the Atlantic is suspected to have pushed the distribution of capelin larvae northwards and the adult westward reducing the overlap in distribution with cod (shown for years 2006-2008 in Pálsson et al. 2012). During this period, the cod consumption of capelin stopped being directly related to capelin abundance (Pálsson and Björnsson 2011). Small and large cod compensated by eating other species but medium size cod appeared more limited in choice. During this period of low access to capelin, cod mean weight-at-age declined by 10-20% (Pálsson and Björnsson 2011). Similarly, the distribution of capelin in the Barents Sea has shifted northward, likely due to water warming (Peck 2011b).

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[2] This is very different from terrestrial ecosystem where for instance, wolves eat deer and moose but none of these prey feed on young wolves.

Current decrease in salinity and ocean warming in the Baltic Sea favours sprat and herring due to an increase in their prey abundance. Concurrently, cod, a sprat and herring predator, suffers from a reduction in habitat and thus, reduced spatial overlap with sprat, and reduced biomass (Peck 2011b). Similarly, variations in forage fish biomass and phenology driven by environmental conditions may impact marine birds. For instance, in Newfoundland, increased water temperature in the late 1990s and the resulting earlier capelin spawning led to a decline in the number of gravid female capelin delivered to the murre chicks (Davoren et al. 2012). Currently, capelin spawning occurs later (July-August) in Newfoundland.

Environmental conditions may have a large influence on recruitment (Pikitch et al. 2012a) by changing prey availability or modifying larvae retention or transport. For instance, weak currents in the North Sea may be detrimental to herring by reducing the transport of larvae to the nursery areas on time and likely caused the low recruitment of the 1970s. Conversely, these same weak currents could have been responsible for retaining larvae of sandeels and sprat on the Shetlands grounds and improving survival (Corten 2013). Concurrent favourable water temperatures could also have fostered increased survival through better feeding conditions (Corten 2013). In upwelling ecosystems, climate is known to determine the type of phytoplankton –cold water favours large diatoms over dinoflagellates- (van der Lingen et al. 2006) which in turns promotes the production of large zooplankton (Alheit and Niquen 2004). Thus, climate regime changes are known to switch the dominance between sardine (warm phase) and anchovy (cool phase) in all upwelling ecosystems (Boyer et al. 2001; Chavez et al. 2003). In the Northwest Pacific, carrying capacity for herring was directly linked with the amount of primary productivity and thus environmental conditions (Perry and Schweigert 2008). In the North Sea, sandeel recruitment seems related to the amount of food consumption and likely the type of food available (small or large zooplankton species; determined by environmental conditions) and density-dependent limitations (Peck 2011a).

Abrupt changes in climate regimes can make fisheries management difficult because of consequent large and unpredictable changes in forage fish biomass. The production of several strong cohorts during a favourable climate regime can result in inflated estimates of stock biomass and lead both industry and fisheries managers to be overly optimistic, believing that the stock is not vulnerable to current and increasing level of exploitation. However, a period of low recruitment caused by poor environmental conditions combined with high levels of exploitation rapidly leads to overfishing and stock collapse as was the case in the northern Benguela system (Namibia) in the 1960-1970s (Boyer et al. 2001). Furthermore, climate anomalies of 1993-1995 (reduced upwelling, warmer water, and intrusion of a body of poorly oxygenated water) led to decreased sardine recruitment, decreased abundance in several other species, and in the displacement of hake and monkfish (Boyer et al. 2001). The ecosystem was modified into an alternate (degraded) state in which dominance changed to jellyfish and pelagic gobies, both non-exploited species (Bakun and Weeks 2006; Bakun et al. 2010). Thus, both the quantity and quality of the forage base declined (Ludynia et al. 2012). This culminated in the dramatic decline in several species such as the Cape fur seal, Bank cormorant, African penguins and Cape gannets (Boyer et al. 2001; Ludynia et al. 2012).

CONTRIBUTION TO ECOSYSTEM AND FISHERIES

In spite of their perceived importance in ecosystems, very few studies have quantified the contribution of forage fish to ecosystem predators, to directed fisheries or to other commercial fisheries. Using 72 Ecopath ecosystem models around the world, Pikitch et al. (2012b) found that predators' dependency on forage species is the highest in Antarctica and lowest in tropical lagoons. Similarly, the total production of predators supported by forage fish varies considerably among study areas, being highest in upwelling, in Antarctica ecosystems ($>9\ t{\cdot}km^{-2}{\cdot}year^{-1}$), and in higher latitudes ($3.79\ t{\cdot}km^{-2}{\cdot}year^{-1}$ vs $1.18\ t{\cdot}km^{-2}{\cdot}year^{-1}$ at low latitudes) (Pikitch et al. 2012b). Considering all forms of contribution, these species were found to contribute a minimum of 20% of the global ex-vessel[3] value of all marine fisheries combined in the world. This exceeds their direct value on the market (Pikitch et al. 2012b).

VULNERABILITY TO FISHING

Although forage fish exhibit a rapid response to environmental conditions suggesting high resilience, there is a documented risk of collapse for these populations (Pinsky et al. 2011), and of modification of the ecosystem structure (Cury et al. 2004). Indeed, their population abundance ranges from large recruitment in favourable conditions to spectacular collapses in unfavourable conditions combined with heavy exploitation (Beverton 1990; Chavez et al. 2003; Pikitch et al. 2012a; Zwolinski and Demer 2012).

The relative contribution of environmental conditions and fishing to population decline is not always clear. For instance, both overfishing and changes in environmental conditions have been proposed as determinants of herring population dynamics and migration behaviour in all Atlantic and Pacific populations (Hay et al. 2001). Although most stocks around the world seem to have recovered from overexploitation, the seriously depleted Hokkaido-Sakhalin stock has not yet recovered in spite of a serious decrease in fishing pressure (Hay et al. 2001). Similarly, in the NW Atlantic herring stocks in SW Nova Scotia have not fully recovered to historical biomass levels (Melvin and Stephenson 2007; Power et al. 2010).

The profound change in ecosystem structure observed in the Black Sea during the 20th Century resulted from "the perfect storm" combining hydroclimatic and several anthropogenic factors spanning overexploitation of large predators, eutrophication brought by agriculture, and the coincidental invasion by alien species of jellyfish (Daskalov et al. 2007). This series of events started with the depletion of top predators in the early 1970s, that resulted in an increase in small pelagics and the dominance of industrial fishery for small pelagics (80% anchovy and 20% of sprat and a small variety of horse mackerel) in the 1980s (Daskalov 2002), increasing their mortality (Daskalov et al. 2007)[4]. During the 1970s, the native zooplanktivorous jellyfish biomass increased, later replaced by the invasive species. Consequently, zooplankton biomass decreased by around 2-fold and phytoplankton blooms became more frequent. The unutilised phytoplankton sank the bottom where it accumulated, resulting in anoxia and mortality of benthic filterers (e.g. mussels), compounding the problem (Daskalov 2002). This describes a perfect trophic cascade. Concurrent increases in forage fish and jellyfish are attributed to different prey selectivity, fish removing the larger zooplankton species, resulting in more small species available for jellyfish. In this ecosystem, energy pathways from plankton were modified towards jellyfish instead of small pelagics, while the food chain was truncated from 4 to 3 trophic levels. Although the ecosystem exhibited a short recovery in the 1980s, it did not last as jellyfish increased again (Daskalov et al. 2007), confirming that the path to recovery, if possible, does not simply consist in the reversal of the path of "destruction".

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[3] value before processing

[4] Fishing mortality increased to more than 2 in the 1980s and at about 0.6 in the early 2000s (figure 1b in Daskalov et al. 2007). The author does not mention any overexploitation for forage fish.

Forage fish are vulnerable to overfishing for several reasons.

First, their propensity to form large schools that are maintained even as abundance declines and locally results in higher catchability with greater risks of collapse in presence of heavy exploitation. A large body of literature documents this phenomenon and consequently how catch per unit effort (CPUE) cannot be used to track changes in abundance (Saville 1980; Csirke 1989).

Second, as with other fishes, exploitation often results in age truncation. In forage fish populations with a short life span, a reduction in number of age classes increases the risk of poor recruitment by reducing the number of older fish, their particularly important fecundity and migration knowledge (McQuinn 1997).

Third, in some forage fishes (e.g. clupeids), recruitment is not always strongly driven by density-dependence and thus recruitment can be highly variable regardless of the spawning biomass. For instance, some of the largest year classes have been produced by very low spawning biomass, yet recruitment has also been found to be consistently low at low adult stock size (menhaden, ASMFC 2010; Scotian shelf herring, Power et al. 2010; e.g. Norwegian herring, Røttingen and Tjemeland 2012).

Fourth, natural mortality (M) is high in these populations due to predation and it is likely variable for the different life stages. For instance, herring are eaten throughout their life but predation on herring is 73% higher on juveniles (<2 years old) than on adults on the Western Scotian Shelf (Guénette and Stephenson 2012). M is often estimated equal to or higher than (even double) the estimate of fishing mortality (Tyrrell et al. 2011, and reference therein; Pikitch et al. 2012a).

Pikitch et al. (2012a) stated that precautionary management is necessary for these species because of the difficulties involved in quantifying forage fish abundance, the large and abrupt variations in abundance and the consequences on dependent predators. In addition, single-species management can lead to collapse when unfavourable environmental conditions and the effects of heavy fishing act simultaneously.

The ultimate precautionary management is to banish fishing of species that are deemed too important to the ecosystem to lose. For instance, the krill fishery was banned in the US before the fishery was even started (Pikitch et al. 2012a). In Canada there is a small, but highly restricted krill fishery on the west coast. The fishery began in 1970 and continues to date with a low harvest level (500 t), fixed number of licences and a limited area of operation. The TAC was set at no more than 3% of estimated krill consumption by all predators in the Strait of Georgia (Everson 2007). However, the fishery is unmonitored and no information is provided for assessment or management purposes. On the east coast of Canada an exploratory fishery for krill and copepods began in 1993 and TAC's set at 100 t and 50 t respectively. The TAC's were increased to 300 and 2000 t in 1995 for krill and Calanus, yet catches were less than 2 t. In the same year a proposal for a 1000 t experimental krill fishery on the Scotian Shelf was submitted to DFO (Head 1997). The review process concluded that a 1000 t fishery might not adversely affect the ecosystem but no fishery was allowed mainly because a successful fishery might want to expand. Also, there was not enough information on the stock to devise credible management rules, in some areas there was an existing herring fishery, and these areas are also important feeding areas for the endangered north Atlantic right whale. In summary Canada's approach to the harvesting of krill has been one of appropriate caution, management restrictions and ecosystem considerations (Everson 2007).

In Alaska, exploitation of forage fish other than herring is prohibited owing to their importance in the ecosystem and the lack of knowledge of their population dynamics. By-catch is limited at 2% of landings in federal waters, (Ormseth 2011)[5]. In these cases, the primary management objective was to protect the species and the ecosystem it supports. The following section outlines several possible approaches for management strategies.

MANAGEMENT STRATEGIES

In most fisheries, the management objectives are defined and include at minimum, yield maximisation but also stock rebuilding, maintaining biological productivity, protecting predator food base, and other ecosystem considerations as discussed above. For instance, the management of the Barents Sea capelin aims at maintaining a spawning biomass large enough to avoid population collapse and avoid starvation for cod, a dependent predator. In Canada, management strategies vary from stock to stock depending on the level of information. In general, they aim at sustaining the resource but use different strategies. The Gulf of St. Lawrence herring is managed to $F_{0.1}$ while in North Atlantic Fisheries Organisation (NAFO) Division 4WX and Newfoundland they are managed based on trends in biomass or other indices.

These management objectives are operationalized by defining reference points and harvest control rules, an ensemble of which are called here **management strategies**. **Reference points** are values derived from analysis (historical data or model-based) describing a state of the population/stock used to compare with current status and inform management decisions. For instance, avoiding overfishing necessitates the definition of a technical point beyond which it is recognized that overfishing is occurring (Caddy and Mahon 1995). Reference points can be values to be achieved (target) or states to avoid (limit). **Harvest control rules** are pre-agreed rules that determine the course of management action as a response to current stock status relative to the reference point.

Management strategies can be divided in four groups depending on the type of data and principles they are based on: 1. empirical indicators and reference points, 2. stock assessment-based reference points, 3. temporal and spatial characteristics, and 4. ecosystem-based. The requirements for data and formal stock assessments increase from empirical to temporal and spatial approaches and from single species to multi-species and ecosystem approaches. These strategies are not mutually exclusive and can be combined. We present a brief overview inspired by the most recent publications and thinking on the subject. Management strategies have not been developed for forage fish specifically; at best they were consistent with that of other species.

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[5] capelin is not part of the list of restricted forage fishes but does not support targeted fishery (Brown 2002)

1. EMPIRICAL INDICATORS AND REFERENCE POINTS

Empirical reference points are derived directly from data to form simple rules to guide the level of harvest. However, simple indicators partitioned by age or size, such as indicators based on age and length structures, the percentage of large spawners (mature fish) or fish larger than the optimal length in the catch (Froese 2004), are not well suited for small pelagics because small pelagics can have widely variable recruitment. Pikitch et al. (2012a) proposed the use of age-specific body size to detect density dependent effects on growth (based on Lorenzen and Enberg 2002). However, this indicator would be difficult to interpret without an index of biomass because environmental conditions may also affect weight at age. For instance, it would be difficult to differentiate between high biomass (high density dependence) and exceptionally poor environmental conditions. The same could be said of other indicators of fish condition maturity, gonadal weight and physiological condition.

Alternative indicators are dependent predators' biomass or reproductive success, which are influenced by forage fish biomass and the availability of forage fish to predators. Marine birds have been shown to be particularly sensitive to changes in food supply and as such could be used as indicators of overfishing (Pikitch et al. 2012a and reference therein). Cury et al. (2011) recommend keeping at least one third of forage fish in the ecosystem to maintain nesting sea bird populations. Below this threshold, breeding success was shown to decrease in 7 ecosystems.

In the Antarctic, CCAMLR adopted ecosystem-based management, restricting the harvest of krill to ensure the presence of a sufficient biomass for predators needs (see the Antarctic krill case study). Testing various indicators of effects of fishing, Reid et al. (2005) found that breeding success was among the more responsive to krill abundance, and that response varied among predators as a function of their level of dependence, foraging range and other factors not taken into account in the study (residuals). Thus, the best predictor was a multivariate combination of factors.

Such indicators require knowledge and understanding of the trophic links, and their strength, between forage fish and their predators. They are based on data derived from monitoring programs for predators and provide information about the effects of fishing that cannot be obtained otherwise, but require additional resources. Further, monitoring predators implies an unavoidable lag between the effective change in forage fish population dynamics and the time the effect on the predator is recorded. Thus, harvest strategies usually rely on direct monitoring of the forage species. The negative effect on predators, when identified is recognition that a lower limit threshold, which should be avoided, has been reached.

2. STOCK ASSESSMENT BASED REFERENCE POINTS

Stock assessments, based on indices of catch, abundance and knowledge of population dynamics are used to assess population status and to derive reference points. A wide variety of modelling methods are used, such as analytical age-structured models (e.g. Virtual Population Analysis, Statistical Catch at Age, stock synthesis etc.) to determine indicators such as age-specific fishing mortality rates and abundance at age (Pikitch et al. 2012a).

Fishing mortality rates

Management strategies that are based on constant fishing mortality (F) result in inter-annual variation in target yield (Total Allowable Catch, TAC) corresponding to a constant fraction of the estimated stock biomass (Figure 2). In the past, constant F was based on F_{MSY}, the fishing mortality that would procure the maximum sustainable yield in the long-term (see Appendix 1). In hindsight F_{MSY} was too optimistic and is now generally interpreted as a limit rate to be avoided rather than a target (Mace 2001). Consequently, several arbitrary rules have been devised to serve as a proxy for F_{MSY} (Clark 1991) or to derive a more cautious fishing rate that would decrease the risk of stock collapse.

Among them, $F_{0.1}$ (defined as the point on the yield versus fishing mortality curve that corresponds to the value of fishing mortality (F) where the slope is one tenth of the slope at the initial slope - see Appendix 1) is used widely in eastern Canada and around the world (Hilborn and Walters 1992) and recently, the European Commission (and ICES) opted to use $F_{0.1}$ as a proxy for F_{MSY} (European Commission 2006). Other constant mortality rates are based on a combination of yield-per-recruit and stochastic modelling and result in a family of target fishing mortality that would reduce the spawning biomass to some fraction of the unfished biomass ($F_{35\%}$ or $F_{50\%}$).

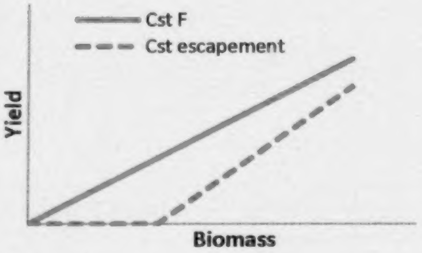

Figure 2. Illustration of control rules featuring constant fishing rate alone or coupled with a limit biomass (constant escapement).

The reference points for F could also be set as a fraction of natural mortality $F=eM$, where M is the natural mortality derived from growth data, and e is set based on meta-analyses or modelling (Walters and Martell 2004). Based on a meta-analysis of exploited small pelagics, Patterson (1992) showed a probability of 50% decrease in stock biomass when a long-term exploitation rate **$E=F/Z=0.4$ ($F\sim 0.67M$)**. Exploitation rate below 0.3 ($F\sim 0.5M$) allowed stocks to increase in size but most stocks declined with an average exploitation rate above 0.5 ($F=M$).

Spawning potential can serve as another guide to determine precautionary fishing rules. Using a yield-per-recruit model, it is possible to calculate the ratio of egg production in a fished population compared to the egg production in an unfished population. To avoid recruitment overfishing the ratio should not be less than 0.3 and to avoid long-term change in community structure the ratio should not be less than 0.5 (Walters and Martell 2004). The target F is the rate that maintains the desired ratio.

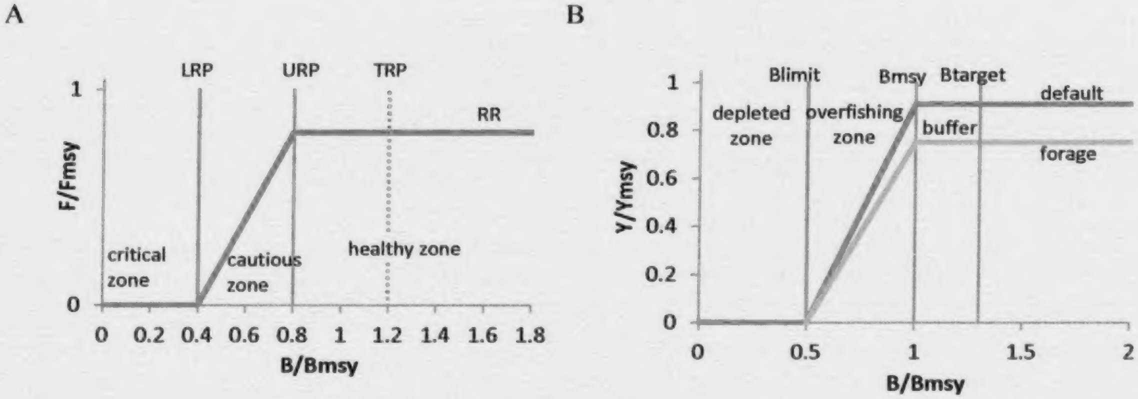

Figure 3. Illustration of control rules featuring A. the Canadian generic harvest strategy (DFO 2009) and B. the generic harvest control rule proposed by Froese et al. (2011) for European waters.

LRP: limit reference point; URP: upper stock reference; TRP: target reference point (undefined and presented for illustrative purposes); RR: removal reference, undefined but always smaller than F_{MSY}. The Canadian policy defines LRP as 0.4 B_{MSY}, and URP as 0.8 B_{MSY};

Harvest control rules

Harvest rates and biomass thresholds can be combined in various ways to maintain a minimum total biomass or spawning potential for the stock. For example, a constant harvest rate could be used while the spawning stock biomass remains higher than a set threshold below which fishing is reduced or suspended (Pikitch et al. 2012a) (constant escapement Figure 3A). Several control rules are formulated with two biomass reference points: the critically low biomass (B_{limit} or LRP in the Canadian policy[6]) and the upper biomass threshold (B_{high} or URP in Canada[7]). There is no fishing when biomass is below the lower limit, and F increases as biomass increases toward B_{high} after which point *F* is set at a fixed level (e.g. Figure 3A, Appendix 2).

Based on a meta-analysis of European stocks, Froese et al. (2011) suggested a default reference point of 0.5 B_{MSY} as the default B_{limit}, and of 1.3 B_{MSY} as a target[8]. The target biomass was chosen based on the 95% confidence limits of the estimated B_{MSY} for the European stocks, to account for the uncertainty in estimating B_{MSY} and give a buffer for biomass fluctuations, but does not take into account the error in current biomass estimation. Note, this involves a fixed TAC when B is above B_{MSY} (Figure 3B) and a decreasing F as the stock biomass increase over B_{target}. The authors suggest a target of 1.5 B_{MSY} (still providing 75% MSY) for forage fish, to account for their role in the ecosystem and the risk of ecosystem erosion when all species are exploited at F_{MSY} (Walters et al. 2005).

Forage Fish Control Rule: Given the large uncertainty in estimation of biomass and the amount of information required for most stock assessments (see above), Pikitch et al. (2012a) proposed a theoretical approach named the Forage Fish Control Rule (FFCR). It is based on the Potential Biological Removal- PBR (Wade 1998), currently used to estimate the maximum mortality, in addition to natural mortality, that marine mammal populations can withstand, and also achieve protection and rebuilding of their populations (see Appendix 3, Box 1).

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[6] LRP: limit reference point in Canadian policy (DFO 2009)

[7] URP: upper stock reference

[8] This will provide a harvest of 0.91 MSY, thus not a great loss in yield

The FFCR sets the harvest (quota) of a fish as a function of the intrinsic rate of growth r, estimate of current biomass B, minimum biomass observed $Bmin$, a conservation factor C (similar to that of the *PBR*), and the number of dependent predators N:
$Quota = max(0, C/(N+1) * 0.5r * (B-Bmin))$

This equation results in a form of variable fishing rate (and yield) above a limit biomass (*Bmin*) and the yield is kept lower than the yield at MSY. The rule accounts for the needs of predators and is intent in considering humans as just another predator that should consume no more than other predators (Appendix 3 Box 2). The FFCR rule results in harvest rates and yields that increase with biomass but they are considerably lower than what would be obtained by a conventional (modest) constant harvest rate set at 0.2 (Figure 4).

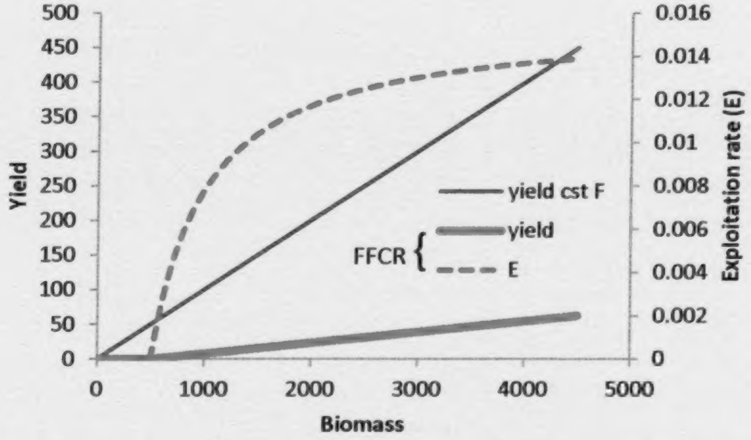

Figure 4. Yield and exploitation rate (E) under the Forage Fish Control Rule (FFCR) compared to the yield obtained with a constant F. Parameters used: r=0.5, C=0.5, N=7, Bmin=500, constant F=0.2.

3. SPATIAL AND TEMPORAL CLOSURES

Spatial and temporal closures[9] are not new ideas and have been commonly used in Canada and around the world. Typically, they limit fishing based on season and/or locations to control catches, bycatch, and fishing mortality or protect part of the life history (e.g. spawning aggregations) (Pikitch et al. 2012a). Their utilisation enables the spatial structure of ecosystems to be taken into consideration, a feature that is not typically included in quota management, and which could play an important role in achieving broader goals of ecosystem-based management (O'Boyle and Jamieson 2006; Olsen et al. 2007; Halpern et al. 2010).

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[9] Here we use closed areas in a generic term that includes MPAs and marine reserves with various objectives and sizes. Typically these areas would exclude, at minima, fishing activities in part or totally.

There is a large body of literature examining the theoretical (Gerber et al. 2003; Gerber et al. 2005; Grüss 2012) and observed effects of closed areas (Halpern 2003; Mumby et al. 2006; Goñi et al. 2008). The key determinant of the success of closed areas is the level of protection provided to the population(s) they are designed to protect. Protection comes in several forms, such as protection of different life stages (spawning, nursery, feeding areas), protection of species movements or migrations, protection of species dependent habitat structure (Babcock et al. 2005; Grüss 2012), protection from fishing., e.g., the level of fishing (Halpern et al. 2010) and the level of redistribution of fishing effort in unprotected areas (e.g. Guénette et al. 2000).

Forage fish could benefit from closed areas that protect juveniles from bycatch or directed fisheries, or protect the core area of the population that features range expansion and contraction as a function of the biomass and environmental conditions. For instance, closed areas are used in the Gulf of California as a tool in the sardine fishery to protect spawning aggregations along with other management measures (Bakun et al. 2010). Species that are tightly linked to a particular habitat, e.g. the North Sea sandeel and Norway pout (see case studies below) are well suited for this type of management.

Closed areas protecting forage fish aggregations could prevent local depletions and consequently, protect dependent predators from decreased access to prey. The key question is the degree of dependence of predators and the risk of local depletion. Several Marine Protected Areas have been implemented to protect critical habitat for marine mammals, but only a few were meant to protect feeding aggregations (Hoyt 2005). There are several areas closed to fishing to avoid local depletions for nesting birds. A large MPA covering 400 km of coastline in Namibia, was meant to protect several components of the ecosystem along with feeding areas for birds (Ludynia et al. 2012). The expected success of this MPA relies on the fact that several species feed mainly within the boundary of the MPA (Ludynia et al. 2012).

4. ECOSYSTEM-BASED MANAGEMENT

The management strategies outlined above were mainly devised for single-species management. The key role of forage fish described in previous sections brings home the need to account for other components of the ecosystem and to protect ecosystem structure. Recently though, governments, international agencies, scientists and environmentalists have called for consideration of ecosystem structure, processes and species interactions when managing fisheries (FAO 2001 ; Pikitch et al. 2004; Babcock et al. 2005; Gavaris 2009). This would allow non-fisheries benefits of marine ecosystems to be maintained and sustain both ecosystems and yield. Ecosystem-based management (EBM) is aimed at preventing overfishing, protecting sensitive species, conserving genetic diversity and structure, and maintaining ecosystem structure and habitats. This involves acquiring knowledge of both natural and anthropogenic processes linking trophic levels within the ecosystem (Livingston et al. 2011).

Internationally a number of major initiatives and workshops have been undertaken to advance an ecosystem approach to management and the important role of forage species to the ecosystem as a whole. The Lenfest Forage Fish Task Force (Pikitch et al. 2012a) concluded that the best management scenarios are those using harvest control rules with 2 reference points (B_{lim} and B_{target}) and that current reference points (F_{MSY} and $F_{0.1}$) may not be sufficient to sustain both forage fish and predators requirements in an ecosystem-based management. Based on simulations with 10 ecosystem models and including the effect of uncertainties, the authors propose that the management of a fishery be more precautionary when knowledge is low and uncertainty is high. In such cases, the authors propose that exploitation of forage fish be severely restricted so that the biomass does not decline below 80% of unfished population (B_0), and that precautionary spatial closures be implemented to protect against localised depletion (similar to what CCAMLR has implemented; see the Antarctic krill case study). In the situation of intermediate information about forage fish status, predators' needs and spatial structure, exploitation could be increased and reference points changed to B_{lim}=0.4 B_0 while F would be less than 0.5 of M. In the best case scenario, B_{lim} should not be lower than 0.3B_0, and the harvest control rule should include a buffer zone to account for uncertainties and unpredictable variations. In all cases, spatial closures were considered as a useful tool. This approach, based on the level of uncertainty in forage species status, would represent a major shift (more precautionary) from the conventional approach to the management of forage fisheries in Canada and most ecosystems in the world.

Reference points derived from single-species techniques do not account for predation mortality, which constitutes the main source of mortality for most forage fish (Tyrrell et al. 2011). The most important reason to include predation mortality in assessments is that a large part of the predation is directed at juveniles so that the traditional use of one *M* value for all ages creates a bias in estimates of numbers at age. However, including more realistic natural mortality patterns in analytical assessments results in higher biomass and lower fishing mortality estimates and thus, the revised reference points (MSY or yield-per-recruit based) are estimated at appreciably higher values (see examples in Collie and Gislason 2001; Tyrrell et al. 2011). The new maximum sustainable yield, higher than that obtained from single-species assessments, should of course not be destined entirely to fishing, as predation needs to be subtracted from the surplus production to obtain the biomass available to fishing (Overholtz et al. 2008). In contrast, ecosystem models including predator dynamics and/or changes in ocean productivity lead to lower estimates of MSY (Walters et al. 2005; Mohn and Chouinard 2007; Tyrrell et al. 2008; Worm et al. 2009; Bundy et al. 2012; Link et al. 2012). Also, the speed of recovery of depleted stocks is expected to be quite slower when predation is taken into account compared with single species projections based on potential stock productivity (Tyrrell et al. 2011).

Using ecosystem models (Smith et al. 2011) found that fishing low trophic-level fish (forage fish) at MSY would result in large impacts in the ecosystems studied. The level of impact depended on the level of abundance and connectivity of the forage fish species. Smith et al. (2011) recommend fishing at a level that would achieve 80% of MSY. Pikitch et al. (2012a) found similar results using 10 ecosystem models; fishing at 75% MSY would result in keeping forage fish biomass close to 88% B_0 or higher and avoid a decrease in dependent predators of more than 50%. This is higher and more conservative than the "1/3 for the birds rule" that was proposed as an empirical reference points to maintain nesting sea bird populations (Cury et al. 2011).

Examples of implementation

Ecosystem-based management is still mostly a fledging undertaking, although progress is being made in Canada and internationally, (Link et al. 2011; Link and Bundy 2012), with many challenges still remaining (Cowan Jr. et al. 2012; Samhouri et al. 2013). In Canada, the "New Ecosystem Science Framework in Support of Integrated Management" [10] has defined 8 priority areas for an Ecosystem Approach, including setting ecosystem objectives, selecting ecosystem indicators, developing a risk-based framework, generating integrated ecosystem information for fisheries management, identifying habitats of special importance, considering impacts on aquatic biodiversity, understanding pathways of effects driving changes and understanding climate variability and impacts on resources (DFO 2007a). Integrated ocean management plans have been developed for large ocean management areas such as the eastern Scotian Shelf (DFO 2007d) and the Pacific North Coast Integrated Management Area[11]. In some ecosystems, such as the Baltic Sea and Alaskan waters, there are also multiple management rules and strategies are in place to integrate different components of the ecosystem. In all of these examples, there are still a lot of unknowns about the dynamics of these ecosystems, but concrete management strategies have been implemented to address the main issues. The experience of the Barents Sea and the Antarctic is presented in following sections.

In the Baltic Sea, anthropogenic impacts (high fishing mortality, seal hunting, and decline in water quality) add constraints to a sensitive ecosystem driven by large fluctuations in salinity (Casini et al. 2011). The influences of climate and trophodynamics have been well studied and the lessons drawn from these are slowly being included in fisheries management. The relationship and dominance regimes between cod, herring and sprat are modified by fisheries and climatic changes (Casini et al. 2008; Möllmann et al. 2008; Mölmann et al. 2009). Several aspects of trophodynamics (plankton structure, level of predation and cannibalism) are already included in assessments and forecasts for this region, but the precautionary reference points devised under single-species management have been abandoned in acknowledgement that they cannot be defined for any one of the three species without considering the state of the other two (Casini et al. 2011). In addition these reference points are no longer valid under the new ecosystem conditions of climatic regime shift and the ensuing change in species dominance. Thus, the multi-annual plan for cod is based on a target fishing mortality (0.3) with ecosystem considerations, a goal that should be reached by gradually reducing the quota and has already produced results (Casini et al. 2011). Fisheries advice is still mainly single-species based with added considerations for changes in system productivity and the relative abundance of other species although a multispecies assessment model is being tested (ICES 2009).

Alaskan waters were heavily exploited at all trophic levels for invertebrates, fish and mammals, until the implementation of the exclusive economic zone and the adoption of the Magnuson-Stevens Fishery Conservation and Management Act in 1976 (Livingston et al. 2011). The Alaskan fisheries management for groundfish and crabs has been gradually adding ecosystem considerations by setting maximum bycatch targets, using conservative/cautious fishing mortalities, and protecting habitat and sensitive species. Information on climate, lower trophic groups (e.g. plankton) and predators (e.g. birds, mammals) are assessed in a dedicated ecosystem status report [12]. F_{MSY} is used as a limit reference point while the target fishing mortality is always lower. For species where the spawner-per-recruit (SPR) relationship is uncertain, a proxy for F_{MSY} is used, $B_{35\%}$, the biomass at which fishing mortality reduces the number of SPR to 35% of the unfished level. In practice, a more cautious limit of $B_{40\%}$ is used when B_{MSY} is not known: fishing rate is reduced below this limit to avoid overfishing[13]. In addition, since 1981, a global groundfish catch of 2 million tonnes is set for the Bering Sea/Aleutian Islands, an amount that has restricted catches, in most years. As a result, some stocks, especially flatfish have been exploited below sustainable levels (Witherell et al. 2000). As a result of the cautious management, only one of the harvested species is currently considered overexploited (Livingston et al. 2011). The protection of sensitive or ecologically important species is achieved by implementing maximum bycatch and vessel monitoring (Livingston et al. 2011). Fishing closures have been used to decrease gear overlap but increasingly to protect critical habitat and life stages. For instance, the very large closed areas to bottom trawling off Southeast Alaska, Kodiak Island and the Bering Sea aimed at protecting benthic habitats and commercial resources (Witherell et al. 2000). The climate influence is taken into account in ecosystem and multispecies modelling along with trophodynamics that are used to draw the big picture and evaluate effects of various climate and fishing scenarios. Climate influences are also included in some assessment models to predict species responses. The robustness of management strategies are starting to be tested using formal management evaluations (Livingston et al. 2011). And, the reporting on forage fish has been expanded to more species and included by-catch, distribution and abundance, and diet related information as a response to increasing evidence of their importance (Ormseth 2012). Except for herring (see section below), forage fish species management is limited to by-catch regulations.

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[10] http://www.dfo-mpo.gc.ca/science/publications/ecosystem/index-eng.htm, accessed 24 – October 2013

[11] http://www.pncima.org/media/documents/pdf/draft-pncima-plan-may-27--2013.pdf, accessed 24 October 2013.

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[12] http://www.afsc.noaa.gov/REFM/stocks/assessments.htm

[13] thus using MSY as a true limit reference points.

Ongoing projects with a focus on forage fish

The EU Forage Fish Interaction (FACTS) program: this program (http://www.facts-project.eu) began in 2010 in response to a reform in the Common Fisheries Policy (CFP) which highlighted three main objectives; responsible and sustainable fisheries and aquaculture activities, an economically viable and competitive fishing industry, and a fair standard of living for those who depend on fishing activities. These are almost identical to DFO's published strategic outcomes identified on its web-site. To achieve these goals a real need was identified to develop tools to evaluate the biological, economic, and social impacts of forage fishery management decisions. Consequently, FACTS a multi-year, multi- institute, and multi-ecosystem program was established to quantify the trophic interactions between forage fish and the prey and predators to quantify ecosystem responses to perturbations from human activities, provide cost-benefit analyses and cost-effectiveness analyses of perturbations on forage fish populations, and to develop case study specific and generic advice for ecosystem-based fisheries management of forage fish populations in European Seas.

The extensive work by the FACTS initiative focused on both top-down and bottom-up control in four European ecosystems (Baltic Sea, North Sea, Barents Sea, Bay of Biscay) (e.g. Peck 2011b; a, mentioned in sections above). Their work culminated in an ICES/PICES Symposium on Forage Fish Interactions held in Nantes, France November 2012 (http://www.facts-project.eu/). Key panel discussions on "What exactly is a healthy ecosystem?" and "Managing Forage Fish: What do we want and why?" were held near the end of the symposium and will be summarized in the proceedings to be released shortly. The following summarizes the recommendations and requirements for forage fish management in an ecosystem context.

- Determine if all populations have the ability to rebound from extremely low stock sizes. What habitats and their connections are key to the survival and reproduction of exploited stocks?
- Determine the appropriate spatial scale for indicators from aggregated measures for effective management at the local or sub-regional level of forage fish stocks in an ecosystem context;
- Improve and develop empirical data streams to detect indirect effects of forage fish and fisheries on predators, ideally in an adaptive management framework;
- Investigate different regulation systems and associated incentives applied to fisheries on forage fish within the context of the ecosystem approach;
- Continue to develop a suite of versatile modelling tools to explore a broad scope of management strategies within an ecosystem context including ecosystem and economic risk;
- Test the robustness of management approaches via biomass set asides, i.e., a quantity of stock that is set aside before implementing an otherwise catch policy, or exploitation rates, including spatial issues as well as how to include those in bio-economic models. Compensation and market implications, and payment for ecosystem services.

Euro-Basin: With its potential effects on the ecosystem and forage species, global climate is an important factor affecting the ecosystem approach to management. The European Basin-scale Analysis, Synthesis and Integration (Euro-Basin) is a multi-country, multi-discipline collaborative program (including Canada and the USA) designed to improve our knowledge of ocean basin scale processes, research and prediction capabilities of climate change and anthropogenic forcing on the population structure of key plankton and fish (forage) in the ecosystem. The program takes a top-down approach in its understanding of variability, potential impacts and feedback on a broad, basin-scale (i.e., the North Atlantic). Most projects are currently underway according to their work packages with scheduled completion near the end of 2014. Details of the project are available on the program's web site (http://www.euro-basin.eu/).

MANAGEMENT OF FORAGE SPECIES AROUND THE WORLD

ANTARCTIC KRILL

Sea-ice and hydrographic conditions are important determinants of the distribution, abundance, movements, and recruitment of krill (*Euphausia superba*) (Constable et al. 2000). This means that the effect of global warming on the extent and duration of ice cover could lower carrying capacity for this species and its dependent predators. Krill biomass varies inter-annually according to surveys, but was relatively high in 2001/02 (Hewitt et al. 2004). Commercial exploitation of krill began in the early 1970s in the Antarctic and reached 528,201 t by 1982 (Constable et al. 2000). Since 1995, total catch fluctuates between 80–200 kt (http://www.ccamlr.org/en/data/statistical-bulletin) and lower than the recommended quota.

The 1988 management policy adopted a target fishing mortality of $F_{0.1}$ but this level was deemed too high to enable recovery of depleted species and too high for some of the exploited finfish (Constable et al. 2000). Based on the development of criteria to better manage the fisheries, the evaluation of management strategies and the interest they raised, CCAMLR (Commission for the Conservation of Antarctic Marine Living Resources) came to endorse the principle of managing for long-term and the "precautionary" criterion to define catch limit and harvest rules for krill (Constable et al. 2000). The main interest was to move from reactive management to adaptive, long-term strategy.

Precaution is the guiding principle in fisheries management in the Antarctic under CCAMLR (Constable 2011). In order to ensure sustainability and orderly development, fisheries are not allowed to expand until the tools for management are developed, approved and a management plan has been developed. A corollary to this is that the higher the uncertainty the lower the catch compared to the potential catch.

For the three species that are actually exploited in the Antarctic (including krill), a series of management measures have been implemented. The effects of fishing krill on its dependent predators are included in the management plan in the following manner:

- The target krill biomass was set at 75% of the median unfished biomass (instead of the 50% that is normally used under single-species MSY) to account for predator needs (Constable 2011);
- The limit krill biomass was set at 20% of the median unfished biomass (Constable 2011);
- In the Weddell Sea (area 48) the allowable catch was set at 4 million tonnes on an estimated biomass of 44.3 million tonnes ($F<0.1$) (Hewitt et al. 2004)[14]. Further, the allowable catch has been capped at 620,000 t, slightly above the historical maximum annual catch in Area 48 to date (CCAMLR 2000) until it is possible to define the allocation in smaller management units that would address the risks of local depletion more directly (CCAMLR 2010);
- Several spatial catch allocations within subdivisions are presently evaluated, taking into account factors such as spatial structure of the populations, density variability, historical catch, the estimated needs of land-based predators and potential local depletion (Hewitt et al. 2004; Hill et al. 2009);
- The krill population is evaluated almost yearly with shipboard acoustics. In addition, the CCAMLR Ecosystem Monitoring Programme (CEMP), was established in 1989 to detect

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[14] Maximal catch of 5.61 million tonnes in more recent documents (CCAMLR 2010).

changes, particularly with respect to krill-dependent predators, and to evaluate whether observed changes are due to krill fishing or environmental factors (see empirical reference points section) (Constable et al. 2000).

The data on both krill and predator's need are still insufficient to decrease the large uncertainties and expand the fishery (Constable 2011). In contrast to other vessels, krill fishing vessels are not required to have a vessel monitoring system (VMS) on board and are not subject to other forms of monitoring, which makes this fishery poorly regulated and difficult to assess (Gascón and Werner 2006). In general, compliance is linked to individual countries commitment to international conventions and good cooperation (Constable et al. 2000). The sad case of the illegal toothfish fishery shows the limitations of cooperation and science-based management procedures (Constable et al. 2000), but there is no indication that this is the case for krill. However, harvest interest is currently limited for this species for economic reasons but this may be challenged in the future due to technological developments in the fishery and the development of new markets for "krill oil" by the pharmaceutical and nutraceutical[15] industry (Murphy and Hofmann 2012).

The CCAMLR experience is a work in progress continually facing new challenges. The organisation has benefited from a relatively low demand for krill, which has allowed development of management strategies ahead of a major expansion of the fishery (Murphy and Hofmann 2012). CCAMLR has shown that it is possible to set conservation objectives by implementing explicit management measures to this effect. It is not necessary to know everything to define and implement management measures and to reach scientific consensus (despite large uncertainties) (Constable et al. 2000). The most salient feature is the demonstrated will to account for the ecosystem needs and the will to implement the management measures albeit imperfectly.

HERRING

Alaska herring

In Alaska, the exploitation rate (catch/biomass) is capped at 20% of the long-term exploitable biomass, which is a lower threshold than is usually used for species with this life history (Funk and Rowell 1995; Woodby et al. 2005). The fishery management is based on quotas that are area specific.

Southeast Alaska: Pacific herring, *Clupea pallasii*, had been exploited since the 1880s in southeast Alaska for reduction and fish meal, but this practice was phased out in the 1960s (Pritchett and Hebert 2008). Southeast Alaska (SEAK) has been supplying most of the bait for Alaska's longline and pot fisheries. Most of the current annual harvest is taken in the spring roe fishery which developed in the 1970s. The spawn on kelp fishery had stopped in the 1960s and resumed in 1990.

Since 1994, the fishery management plan stipulates that 1. all stocks be duly identified on a spawning area basis; 2. that the mature biomass be assessed before allowing any fishing; and 3. that a biomass threshold be determined for each stock. In Southeast Alaska, the exploitation rate is restricted further to 10 to 20% when the spawning biomass is above the minimal biomass threshold (needed to maintain biological productivity and sustained yield) and reaches 20% when the biomass is 6 times the threshold level (Thynes et al. 2012). Only adult aggregations equal or larger than 2000 t can be exploited at a rate of 10% (quota = 200 t) (Pritchett and Hebert 2008). There are 5 major discrete groups of herring that sustain the commercial fishery[16]. The biomass is estimated using an egg deposit survey, which encompasses about 75% of the egg deposition identified by airplane in 2002. The remaining spawning deposition is rather marginal, often shallow and less dense than the main aggregation, and may account for about 10% of the total egg deposition in SEAK.

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[15] from the words "nutrition" and "pharmaceutical".

Historically, herring were surveyed using egg deposition dive surveys and vessel hydro-acoustic surveys. Acoustic surveys have not been used since 1994 because the method is considered less reliable than egg deposition. The key component of current assessment, the egg-deposition survey, is used to estimate the biomass of mature herring that spawned that season. Two methods of stock assessment are used depending on data available. Since 1994, the method of forecasting herring abundance for major spawning aggregations in Southeast Alaska is based on Age Structured Analysis (ASA) also called stock synthesis or statistical catch-at-age models. This is an age-structured model that relies on time series of abundance data, estimates of recruitment, age, growth, maturity, natural mortality, weight-at-age and spawning escapement (Pritchett and Hebert 2008). In the case of spawning aggregations for which there is no time series, the biomass accounting method is used to project the following year's return of mature herring. It uses the spawn deposition (spawning biomass), age composition of the catch, weight-at-age, and fecundity. The median historical proportion of mature age 3 herring of each stock is used to forecast age 3 recruitment to the spawning biomass (Carlile et al. 1996; Hebert 2012; Thynes et al. 2012).

Bering Sea- Bristol Bay: In the Bristol Bay herring fishery, the largest in Alaska, the limit biomass was first set at 25% of the average biomass estimates (from aerial surveys) observed between 1978 and 1985. Based on modelling simulations, Funk and Rowell (1995) re-evaluated the biomass that correspond to the 25% threshold and found that the biomass used based on historical data was too low owing to bad environmental conditions during the survey. Based on stochastic modelling, they suggested raising the threshold to provide better protection for the stock and faster recovery in case the stocks falls to dangerously low levels while still allowing a good yield. In addition, the authors show that periodic strong recruitment (every 8 to 12 years) sustains the fishery for almost 10 years.

Norwegian spring-spawning herring

Several papers have been written on the 1970 collapse of the Norwegian herring (*Clupea harengus*) stock that was attributed to both overfishing (including large amounts of juveniles) and unfavourable environmental conditions (Tjelmeland and Røttingen 2009). The fishery expanded dramatically from a near-shore fishery in the fjord in the 1940s to a large international offshore fishery carried out by large heavily equipped vessels with sophisticated instrumentation (Røttingen and Tjemeland 2012).

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[16] Sitka Sound, Seymour Canal, Tenakee Inlet, Craig, Kah Shakes/Cat Islands, all of which are considered as discrete stocks for management purposes.

The stock assessment is a VPA performed with the software TASACS (ICES 2012b). The model combines catch-at-age and a variety of fishery independent data sources: 6 acoustic-surveys, tagging data, larval abundance (as a proxy for the spawning stock), and 0-group trawl-survey data. It is worth noting that M was assumed to be 0.15 for adults and 0.9 for juveniles (1-2 years). In the new analytical models for assessment, M is set at 0.5 in the + group (age 15) (ICES 2012c).

After the collapse and the following moratorium a series of stringent management measures were implemented: a minimum size of 25 cm, and low TACs (Tjelmeland and Røttingen 2009). The recovery was facilitated by an exceptionally large cohort in 1983, increasing the spawning stock biomass (SSB) from 0.5 to 4 Mt. When the biomass exceeded the minimal biomass (2.5 Mt), the fishery was allowed to expand while the fishing mortality was kept at $F<0.05$. Harvest rules underwent a long development phase within ICES where maximum mortality and an operational catch ceiling were considered based on 10 year time horizon predictions (Tjelmeland and Røttingen 2009).

Since 2002, the management plan includes 2 biomass reference points. The minimal biomass 2.5 Mt (B_{limit}, based on observed stock recruitment relationship) that should be avoided, and the upper biomass limit (B_{pa}) set at 5 Mt that accounts for uncertainty and act as a trigger. Below B_{pa}, fishing mortality should be reduced from 0.125 to ensure a rapid recovery. Over B_{pa}, quotas would be set using $F<0.125$ (F_{target}). Based on simulations[17], this harvest control rule constitutes a more cautious management than the MSY approach ($F_{MSY}=0.15$) and is predicted to lead to higher long-term catches and lower risks compared to fixed F (Tjelmeland and Røttingen 2009).

Nevertheless, for several years in the last 20, the realised F was larger than prescribed by the management plan (0.125) and F_{MSY}, but mostly lower than 0.2 (ICES 2012b). The biomass increased until 2009, and has since decreased. Owing to low recruitment, the stock is predicted to decrease below B_{pa} even without fishing in 2013 so the quota recommended was estimated at a maximum of 615,000 t down from 883 kt recommended in 2012 leading to a predicted biomass slightly below B_{pa} (4.3 Mt) (ICES 2012b).

In parallel to the implementation of the harvest control rule, Norway has pursued a program to decrease overcapacity and increase fishery monitoring and surveillance (Gullestad et al. 2012). In addition, ecosystem preoccupation has translated into changes in survey structure from fisheries and oceanographic research to ecosystem monitoring. The distribution of mackerel further north, leads to an increasing distributional overlap between herring and mackerel. The latter constitutes both a predator and a competitor for plankton and may outcompete herring (ICES 2012c).

North Sea herring

The North Sea autumn-spawning herring (*Clupea harengus*) population declined markedly in the late 1960s and started rebuilding in the early 1980s. The population is composed of several discrete spawning components with distinct surveys that allow their respective production to be tracked. However, they are managed as a single stock (ICES 2012a, Appendix 3). The age-structured analytical assessment is fitted to several sampling programs. The catch is sampled to obtain catch-at-age and weight-at-age, although the degree of sampling is decreasing (ICES 2012a). Acoustic surveys are used to assess the adult phase of the population and obtain weight-at-age and fecundity. The International bottom trawl survey (IBTS) is used to evaluate the abundance of 0 and 1 year old or recruitment index. Analysis of the IBTS data concluded that the abundance index for adults was poor since it was unable to track cohorts' abundance (ICES 2012a, appendix 3). The new state-space assessment model now includes (among others) age-varying predation mortality obtained from the multispecies model in which it appears that cod and saithe are the main source of mortality for herring (ICES 2012a).

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[17] Based on simulations carried out with 2 models and various assumptions about recruitment.

The harvest control rule is composed of two biomass thresholds, B_{lim} (0.8 Mt, based on observed stock recruit relationship) and B_{pa} (1.3 Mt). B_{pa}, the upper biomass limit, was determined from modelling (ICES 2012a, Appendix 3). When the biomass is above B_{pa}, the target F is set at: $F<0.05$ for juveniles (age 0-1) and $F<0.25$ for adults (age 2-6). Below B_{pa}, F decreases linearly with the estimated biomass[18]. Fishing is still allowed below B_{lim} but at a lower rate ($F_{adult} = 0.1$ and $F_{juv} = 0.04$). In addition, annual changes in quota are constrained to 15%; discards and slippage[19] is banned. The working group recommended that the actual management strategy be fully evaluated (MSE) for robustness.

According to ICES advice report, the stock is considered at full reproductive capacity, the biomass is higher than the reference points and fishing mortality lower than reference points (ICES 2012f). However, this is based on the new stock assessment methodology that includes predation mortality which inevitably increases the estimated total biomass and decreases F estimates. Nevertheless, these new estimates were still compared to the reference points estimated using the oldest assessment data, resulting in optimistic evaluation of stock status. Thus, in spite of the current low productivity regime, ICES recommended to increase the TAC by 15% for 2013 (ICES 2012f). In comparison, the old assessment with its set of calculated reference points would probably be less optimistic. The management plan was over ruled in 2012 when the TAC was almost doubled in spite of the stipulation that increases in TAC could not exceed 15% in a year (ICES 2012a; Naver 2012). Catches for herring were 73% higher than the TAC in 2010 (ICES 2012a).

Gulf of Maine- Georges Bank herring

Herring (*Clupea harengus*) from the Gulf of Maine and Georges Bank are managed as a single stock by the US National Marine Fisheries Service (NMFS). A number of recent studies support this practice, suggesting that they constitute a single stock based on genetic evidence and in spite of morphometric evidence (Overholtz 2006). In addition, the two groups mix extensively. In the fall, herring migrate southward along the southern New England and Mid-Atlantic coast.

Herring in the Gulf of Maine collapsed in 1977 following an intensive period of fishing by distant-water fleets culminating in a reported catch of 470 kt (1968) and a fishing mortality reaching 0.7 (Overholtz 2006). With a population decrease of 90%, offshore fishing ceased (concurrent with the establishment of the 200 mile limit) and the catches, although low, were concentrated in the Gulf of Maine (Overholtz 2002). The stock complex (Nantucket shoals, Gulf of Maine and George Bank) began rebuilding in the 1980s (Overholtz 2002) and has now reached levels similar to before the collapse (Melvin et al. 1996; Overholtz 2006). A portion of the catch from this stock is taken by the Canadian weir fishery in New Brunswick. Total weir catches although variable, declined over the last decade to reach the lowest level in the history of the fishery in 2012 (Northeast Fisheries Center 2012; DFO 2013).

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[18] $F2\text{-}6 = 0.25-(0.15*(1500-SSB)/700)$

[19] Slippage is opening the fishing gear in the water to let go of the unwanted fish. This practice results in massive mortality of the captured fish when it is done after fish are pursed for pumping.

From 1968 to 1998, herring distribution contracted during the population collapse and expanded as their abundance increased and the spawning components were rebuilt and historical distribution patterns re-established when the stock recovered (Overholtz 2002). The age structure expanded starting in the 1990s as the strong 1983 year-class moved through the population (Overholtz 2006). Contraction and expansion of adult and larval distribution has been documented only for the Georges Bank component (Melvin and Stephenson 2007).

Assessment of the stock complex is performed with ASAP[20] and uses several available abundance indices (Northeast Fisheries Center 2012)[21]. The bottom spring and fall surveys have been conducted since the 1960s without interruption, and the winter survey was conducted between 1992 and 2010. In addition, the summer shrimp survey conducted since 1983 was used in recent assessments as an index of the entire population, mixed at this time of the year. Natural mortality used in the assessment model is based on theoretical (Lorenzen model) calculations increased by 50% for years 1996-2011 which was consistent with herring consumption and reduced the retrospective pattern in spawning stock biomass. This new model carried out in 2012 led to less retrospective pattern than that of 2009 but also modified the estimate reference points.

In 2011, the biomass was estimated at 518 kt, the catch reached 85 t and F at age 5 was 0.138, lower than over the last 10 years during which F was increasing and averaged 0.231 (Northeast Fisheries Center 2012). According to the newly estimated reference points (F_{MSY}=0.267, SSB_{MSY}=157 kt, MSY=53 kt) and the indicator of overfished stock being ½ SSB_{MSY}, the stock in not considered overfished. The model that included predation consumption as an additional fleet suffered from the large uncertainty in predators' selectivity (sampling bias mainly) and was characterised by wide variations in natural mortality estimates as well as retrospective error, and was not used. Also, the reference points based on this model resulted in dramatically different values (Northeast Fisheries Center 2012). The working group concluded that reference points were highly sensitive and uncertain.

Recommendations seem to have been followed by the New England Fisheries Management Council and compliance by the sector is rather good judging from recommended quotas that follow scientific advice[22]. As a rule, landings that exceed the quota are removed from the quota of subsequent years[23]. Ecosystem considerations were discussed by the Science and Statistical Committee at the stage of making quota recommendations (Scientific and Statistical Committee 2012). They concluded that standard fisheries reference points are inappropriate for a forage fish but that the present quota setting would meet the ecosystem-based management requirements (undefined). Given that the recommended allowable biological catch (ABC) for 2013 was set at 130 kt with F=0.2, and the total consumption averaged 161,305 t over the entire time series, the committee argued that their recommendations will likely lead to fishing mortality rates well below the natural mortality rate and a stock size above the biomass target with prospects of good size cohorts in recent years.

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[20] Age Structured Assessment Program that uses forward calculations (http://nft.nefsc.noaa.gov/ASAP.html)

[21] Data on the assessment and reference points are taken from (Northeast Fisheries Center 2012) unless otherwise noted.

[22] http://www.nefmc.org/herring/index.html

[23] see http://www.ecfr.gov/cgi-bin/text-idx?c=ecfr&SID=970b0948bce26f8c2260f7ab242d7ec3&rgn=div6&view=text&node=50:12.0.1.1.6.11&idno=50 ; and http://nefmc.org/herring/index.html

SARDINE AND ANCHOVY

South Africa sardine and anchovy

The Southern Benguela upwelling system supports a productive fish community, and is well known for its sardine/anchovy and hake fisheries. In the 1950s, the fishery was dominated by sardine and horse mackerel until the mid-1960s decline and the fishery converted to smaller-mesh nets to target anchovy. This fishery is based on 0+ recruits and thus, it is not possible to forecast recruitment. Also, the management of sardine and anchovy fisheries cannot be optimised separately since juvenile (0+) sardine shoal with anchovy for a large part of the year, leading to potentially large bycatch of sardine in the anchovy fishery. Adult sardines are caught by a directed fishery and as a by-catch of the round herring fishery. Since the 1980s, sardine have been managed conservatively to allow rebuilding of the stock, and a high biomass has been observed since the late 1990s.

The information on the stocks is obtained from two acoustic surveys a year, one for adult in November and one of recruitment in May/June (De Oliveira and Butterworth 2004). There is no analytical assessment for these species although work is being done on the subject. The management plan includes a TAC for directed sardine fishery and a TAB (total allowable by-catch) for sardine. In addition, supplementary fishing for anchovy was added in August/ September, when anchovy and sardine stop shoaling together, for a “clean” fishing season. Both TAC and TAB are revised twice during the year, based on surveys and landings (De Oliveira and Butterworth 2004; de Moor et al. 2011). The quotas calculated with fixed mortality (modified at the low and high ends of abundances) and are subjected to maximum and minimum constraints (de Moor et al. 2011).

The South Africa small pelagics fishery has been managed using operational management procedures (OMP)[24] since the early 1990s, and the first joint sardine and anchovy OMP was implemented in 1994 (De Oliveira et al., 1998). OMP test management scenarios (set of rules) under considerations to provide resource management recommendations (de Moor et al. 2011) and are carried out in collaboration with industry, acknowledging the trade-offs necessary to manage both species (De Oliveira and Butterworth 2004). The performance is evaluated based on management objectives, risks and required trades-offs. Unfortunately, the early plans did not include the possible drastic changes in environmental regimes that lead to switches in relative abundance of sardine and anchovy. As a result, it was impossible to plan for long-term and to apply the management procedure when the switch occurred. Recent large sardine recruitment resulted in a higher carrying capacity than in the past, and led to questioning the management rules, again. Given the boom and bust nature of this ecosystem the management plan has been modified to accommodate sudden changes in regimes translated into opportunities (boom phase) and the large risk of decline (bust) that follows (de Moor et al. 2011). Although sardine abundance fluctuates widely the population seems to have increased due to a recent strong cohort (Peacock 2011; Oceana Group Limited 2012). Compliance is ensured by a system of monitoring of landings and boat activity (vessel monitoring system) (Peacock 2011).

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[24] Management procedures provide a framework to define management objectives, long-term strategy, and guide the decision making while facing inevitable uncertainty and conflicting interests. It consists of a pre-agreed set of rules that specifies the regulatory mechanisms, the necessary data to be collected and their analysis, the indicators of performance and status, and the reference points that will trigger modifications of fishing rules for the near future. The procedure is generally tested for robustness with management simulations that includes all types of uncertainty (management strategy evaluation) and re-assessed every 3-5 years (Cochrane et al. 1998).

The development of the operating management plan for sardine and anchovies has been long and includes an increasing number of rules, reference points and considerations as more was learned about the stocks and their dynamics. Uncertainty had a perverse effect in shedding doubt on the whole procedure for the industry which makes it easier to apply political pressure (Cochrane et al. 1998). During the management plan development (and perhaps even now), scientific recommendations have been overruled or adjusted for socio-economic reasons which could be considered as a failure of management procedures (Cochrane et al. 1998). The process was found useful to force discussions between parties. Nevertheless, although there was acceptance of the value of the process and the interest for greater transparency, the rules were not adhered to when hard decisions detrimental to fisheries were required (Cochrane et al. 1998). Nowadays, their management OMPs play a central role in several fisheries in South Africa (Plagányi et al. 2007) and are well accepted by the fishing industry for its clear process and the low probability of undue influences biasing the outcomes (D. Butterworth, University of Cape Town, South Africa, pers. comm., 30 March 2013). Nevertheless, disputes on the difficult subject of individual allocations or post-apartheid redistribution are not prevented by management procedure any more than it would be in other systems, and led to several litigations in South Africa (Butterworth et al. 2012).

Eastern Pacific sardine (northern stock)

The eastern Pacific sardine, *Sardinops sagax caerula*, was the most important fishery in the western hemisphere in the 1930-1940s (Zwolinski and Demer 2012). The fishery collapsed in the early 1950s under a combination of the onset of a cold period[25] , intense exploitation (rate >20%), and the decrease of the population biomass to below a critical threshold (740 kt) (Zwolinski and Demer 2012). The collapse was characterised by a severe truncation of the age structure, a halt in feeding migrations along the coast, and a change in dominant species and their schooling behaviour. When their biomass became too low, sardines increasingly mixed with other more abundant pelagic species that thrived under the new colder oceanic regime e.g. jack mackerel, anchovy, Pacific mackerel.

This sardine population is composed of 3 sub-populations, but this section only concerns the northern stock ranging from northern Baja California to Alaska. The centre of abundance is located in California. Migrations to the northern end of the range are seasonal; sardines are present on the Canadian coast in summer and return to California in the fall (Hill et al. 2011)[26]. Sardine migrations also vary as a function of abundance and water temperature, both of which promote extensive migrations, notably to the North (Zwolinski and Demer 2012). In recent decades, the stock has resumed its presence in Oregon, Washington, British Columbia and distant offshore California. Spawning in Canadian waters is uncertain and intermittent. The seasonal distribution of sardines in B.C. waters corresponds with the foraging and migrating patterns of a variety of marine fish, mammals, and seabirds including salmon and humpback whales.

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[25] described by the Pacific Decadal Oscillation and the weakening of the California Current (Chavez et al. 2003)

[26] This section is based on the stock assessment document (Hill et al. 2011) unless otherwise noted.

The species may live up to 15 years although typically sardines are less than 7 years old in the catch. Recruitment is highly variable and strongly influenced by environmental conditions. The fishery declined to very low levels in the 1970s, and they were rarely caught in the early 1980s. As the stock increased in abundance, the fishery resumed. In Canada, their extremely variable abundance led to their designation as a species of "Special Concern" by COSEWIC[27] between 1987 and 2002. In 2002 a commercial fishery was opened and a total allowable catch (TAC) established. The Canadian fishery was re-established after the species was not considered at risk (Flostrand et al. 2011).

The northern stock is assessed as a whole using a stock synthesis model that includes fishery and survey data (1993-2011), for each season (Jul-Dec and Jan-Jun) and four indices of abundance: daily egg-production and total egg production estimates of spawning biomass off California (1994-2011); aerial survey estimates of biomass off Oregon and Washington (2009-2011), and acoustic estimates of biomass from California to Washington (2006-2011). Two other data sets are considered but not used: the DFO survey conducted in summer on the west coast of Vancouver Island, and the US (SWFSC) mid-water trawl survey for juvenile rockfish. The Canadian Survey is known to have large CVs indicating large variability and uncertainty (Flostrand et al. 2011). The model also assumes fixed M (0.4), a Ricker stock-recruitment curve for which the parameters were estimated in the assessment model. The model predicts a higher natural mortality than the initial value used.

The objective for the US harvest guideline is to maintain relatively high and stable catch over the long-term and prevent overexploitation. It is based on an escapement biomass, the proportion of the habitat where sardine is found (a function of average sea-surface temperature) and the average portion of the species distribution assumed to be in US waters (87%) vs Mexican waters (13%) while Canadian waters are not accounted for (Flostrand et al. 2011).

$$\textit{harvest guideline}_y = (\textit{Biom}_{y-1}\textit{-escapement}) * \textit{fraction habitat} * \textit{distribution}$$

The escapement level is 150 kt. The habitat fraction can also be replaced by F_{MSY} constrained between 5 and 15% (Flostrand et al. 2011).

The sardine fishery in Canada is managed based on a series of Fishery Management Framework harvest control rules for establishing a maximum annual commercial harvest. Consideration is given to:
1) the current US sardine biomass estimate from the Northeast Pacific sardine stock assessment 2) an estimated running average seasonal migration rate into B.C. waters, and, 3) an annual harvest rate approximating the U.S. rate of 15% since 2002. The realised harvest rate (0.1-3.7%) and catches (8-86% of the TACs) was generally lower in years 2006-2010 (but increasing),

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[27] COSEWIC – Committee on the Status of Endangered Wildlife in Canada

reflecting the low effort compare to the TAC (Flostrand et al. 2011). The harvest rules in each country are not integrated and do not take account one of the other's rules. The level and intensity of migration into Canada varies annually for reasons that are not well understood (Flostrand et al. 2011). Simulations show that depending on the level of migrations into Canada, the combined harvest rates could exceed the targeted 15% (Flostrand et al. 2011). Recently it has been recommended that alternative control rules or biomass forecasting methods be considered for BC sardines given the uncertainty in biomass estimates. Although sardines are acknowledged as an important forage species to several large predators on the west coast, no specific allowance or consideration is applied for this link in the ecosystem within the current decision rules or guidelines.

Based on oceanographic conditions and the current status of sardine, Zwolinski and Demer (2012) forecast a risk of collapse. They observed that oceanographic conditions are presently not favourable at the scale of the Pacific Basin, a context similar to that of the 1930s. In addition, the exploitation rate for sardine keeps increasing (but still below 15%, Hill et al. 2011), the age structure is truncated and, based on survey estimates, the biomass decreased since 2007 and is now lower than the biomass threshold (740 kt)[28] under which schooling and migration behaviour changes. They note that pre-spawning condition has deteriorated, recruitment was low for years 2004 and 2008, and the abundance of other small pelagics (especially Pacific mackerel) seems to be increasing to the point that sardine may not be the dominant pelagic species anymore. Monospecific schools of sardine are decreasing[29]. The authors argue that the use of the habitat size based on temperature in the calculation of allowable catch is now obsolete because its link with productivity was valid in the precedent warm period but not in the current unstable conditions.

Humboldt upwelling anchovy (Peru)

Anchovies (*Engraulis ringens*) live about 3-4 years and can reach maturity within their first year (11 months). They are characterised by high productivity and natural mortality (M=0.8). From the 1950s to the early 1970s, which coincides with an El Niño event (Pauly et al. 1987; Chavez et al. 2003), Peruvian anchovy was the largest single species abundance in the world. Its stock size increased again starting in the mid-1980s with the occurrence of another cool phase (Chavez et al. 2003). These inter-decadal variations in abundance existed long before the onset of fisheries, as determined by paleontological records (Fréon et al. 2008). Landings ranged between 5.1 and 9.1 Mt between 1999 and 2009 [30]. The fishery was first carried out exclusively with an industrial fleet of over 1200 steel boats with 30-900 t of holding capacity, but since 1999 a semi-artisanal fleet of wooden boats with holding capacity of 30-110 t has evolved (Fréon et al. 2008). The number of boats is similar in both fleets but the industrial fleet lands 85% of the catch. The fishing capacity more than doubled between 1987 and 2006, a period of increasing anchovy abundance (see Aranda 2009 for the background). Thus, the overcapacity in fishing and processing capacity was at 37% in the early 1980s increased to 69-75% in 2005.

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[28] The biomass estimate resulting from the analytical assessment, 988 kt, is higher than the threshold (Zwolinski and Demer 2012). It is interesting that their threshold (740 kt) is quite a bit higher than the escapement level used in the harvest control rule (150 kt).

[29] This is called the **school trap.** It is detrimental because it fragments the adult population, disrupts migration routes, foraging and thus reduce reproductive potential.

[30] http://www.imarpe.pe/imarpe/archivos/reportes/imarpe_indust_anchoveta.pdf; http://www.imarpe.pe/imarpe/archivos/informes/imarpe_infpel_rep_pelag_anual_2008_09.pdf

To stop the race for fish, the government of Peru instituted maximum capture limits per vessel (IVQ, Individual vessel quota with 10 years rights to non-transferable quotas) (Aranda 2009). Since then, the mean number of fishing boats at sea per day diminished from 836 to 280, and the number of days-fished increased from 33 in 2008's first fishing season, to 102 in 2009's first fishing season which could mean that the race-for-fish is ending (Sustainable Fisheries Partnership). Vessels are monitored with satellite tracking devices, and catch monitoring is performed at landings sites. This should ensure that quotas are respected but the effectiveness of the controls are uncertain because of incomplete reliability of landing controls and subsequent possible illegal and unreported fishing (Sustainable Fisheries Partnership). The production was entirely used for fishmeal and oil until recently when landings from the semi-artisanal fishery started to be sold for human consumption.

Acoustic and ichthyoplankton surveys are performed at the beginning of each season to estimate spawning biomass, recruitment and age structure of the stock (Fréon et al. 2008). During the surveys, other components of the ecosystem such as predators (birds, sea lions, seals, and other fishes) are monitored (Pauly and Tsukayama 1987; Fréon et al. 2008). Nevertheless, there is no provision for the ecosystem role of this forage fish in the current management scheme and the fishing rate remains high, appearing sustainable for the fishery because of good environmental conditions. Ecosystem modelling shows that high fishing mortality (0.8-1.4) leads to decrease in dependent predators such as birds and mammals (Tam et al. 2010). The probable competition for anchovy between marine birds and the fishery may lead to local depletion and Bertrand et al. (2010) recommended exploring the use of closed areas to protect foraging areas.

Total biomass trends estimated from a VPA vary widely since the 1960s, reaching a peak of about 20 Mt in the early 1970s and the mid-1990s. In 2003-2004, the biomass was estimated at 10 Mt (Fréon et al. 2008). Three types of analytical model have been used for this species: a catch-at-age analysis (Deriso et al. 1985) published by Díaz et al. (2010), a VPA (latest version in Fréon et al. 2008), and a cohort analysis called integrated assessment model that includes calculation of stochastic growth in length (based on von Bertalanffy equation), natural mortality and monthly time steps (Oliveros-Ramos et al. 2010)[31]. The three models result in conflicting trends for the last 20 years, with the VPAs predicting the highest biomass[32]. According to the cohort analysis and the integrated assessment, the biomass in 2007 was 7.5-9.5 Mt. The trends in fishing mortality also differ, indicating fluctuations between 0.6 and 1.4 with the catch at age analysis or 0.5-1.3 with the integrated assessment. The summary of the review panel conclusions state that F has been maintained between 0.6 and 0.8 while exploitation rates were kept below 0.4 (Guevara-Carrasco et al. 2010). Given the past decadal fluctuations in climate, the panel concluded that the favourable conditions for anchovies should remain until about 2020 although the current rate of increase in abundance may not be sustainable (Guevara-Carrasco et al. 2010).

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[31] From the information available in this paper, the model requires the estimation of 309 parameters, including the initial recruitment for year 1, natural mortality (range from 0.6-1.0), length at recruitment for each cohort.

[32] In both publications (Díaz et al. 2010; Oliveros-Ramos et al. 2010) the authors compared their model with a VPA analysis which differ appreciably in trends among themselves. The VPAs used are not documented.

There is no formal control rule but the management aims at maintaining a spawning biomass (B_{pa}) above 5 Mt (based on a 40 year stock-recruitment relationship) at the beginning of each season by using several tactics (Sustainable Fisheries Partnership ; PRODUCE 2011).

- Quotas are established for each of the two fishing seasons;
- Fishing mortality should not exceed natural mortality (M estimated at 0.8 to 1.25);
- TAC should not exceed 40% of the total biomass estimated for a period. In warm years, less favourable to anchovy, the percentage decreases to 33% (in 2011 for instance, the quota was set at about 3.5-3.8 Mt of a biomass of 10.5 Mt, IMARPE 2011);
- TAC should be kept between 5 and 8.5 Mt;
- Juvenile ratio should not be higher than 10% of the catch in numbers or the area will be closed to fishing (as it was in January 2011, Anonymous 2011) and a minimum size of 12 cm is enforced
- Maximum 5% bycatch of other species;
- Seasonal closure to protect spawning peaks, January-March and July-October;
- Spatial closure (fishing operations off 5 nautical miles from the coast);
- Landings from artisanal fleets only for human consumption;
- Effort control (one trip per day, satellite positioning system on board).

According to SFP (Sustainable Fisheries Partnership), TACs have always been set in line with the scientific advice and the TAC has followed the spawning stock biomass since 2007.

BARENTS SEA CAPELIN

Capelin (*Mallotus villosus*) matures at about 3-5 years and dies shortly after spawning. The capelin fishery in the Barents Sea started in the early 1900s but really increased after the collapse of the Norwegian spring-spawning herring in the 1960s (Gjøsaeter et al. 2012)[33]. Joint USSR-Norway management rules were developed in the late 1970s and included a TAC based on acoustic surveys (performed since 1972) and a strategy to preserve a minimum of 500 kt of capelin (based on observed relationships between SSB and recruitment). This escapement strategy requires knowledge of cod consumption and its inclusion in the assessment model. The output of this model determined the natural mortality attributed to predation, and the best season for fishing (winter) to avoid depletion before cod feeding.

The 1984-1988 capelin collapse (the main forage fish in the ecosystem) triggered a new interest for multispecies dynamics as it became clear that a realistic fisheries management plan could not ignore interactions between herring, cod and capelin (Gjøsaeter et al. 2012). This collapse and subsequent ones were caused by low recruitment and increased predation mortality from big herring cohorts entering the Barents Sea. In addition, the exploitation rate was quite high during 1973-1984 (33% and even 73%, Ushakov and Prozorkevich 2002).

The fishery was closed between 1986 and 1990 following scientific recommendations. It was brought to light that the model described above had underestimated the predation pressure of cod on capelin and a cod-stomach sampling program was started in 1984. A multispecies model was built, including cod and marine mammal predation, but was deemed too complicated and data-demanding to be used in its full version, although it was used to determine quotas when the stock size increased again. Unfortunately, the stock increase was based on a single large cohort followed by a series of low recruitment years which triggered a ban on fishing in 1992. The new "multispecies" model used since 1998 includes the impact of herring and a probabilistic assessment. It is still fundamentally a single-species model in which input from other species are included in a uni-directional manner without endogenous dynamics for the predators (e.g. effect of capelin abundance on cod growth) (Gjøsaeter et al. 2012).

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[33] This section is inspired by Gjøsaeter et al. (2012) unless otherwise noted.

The current assessment is based on one acoustic survey (in September), the output from a multispecies model (Bifrost) from which the amount of predation by cod is obtained and used as input in a model that calculates the catch quota. The quota is set based on short-term projections, performed with scenarios including uncertainty, and according to the harvest control rule (Anonymous 2009). Cod consumption in February and March is estimated from stomach content and evacuation rates sampled annually, cod abundance in February (from a separate assessment) and the abundance of capelin. The degree of capelin maturation is modelled as a function of length annually. In addition, prediction of herring abundance in the Barents Sea (from separate assessment) is included in the Bifrost model (Anonymous 2009).

The harvest control rule is designed to ensure that when the fishery is closed, the SSB remains above the proposed B_{lim} of 0.2 Mt (with 95% probability) (ICES 2011a). This is based on the lowest biomass (1989 SSB) that produced a good year class, following the precautionary approach. Because of the strong natural fluctuations, the probability of obtaining a low biomass never drops below 0.2, even when spawning biomass reaches 1.2 Mt in simulations (Anonymous 2009). The target SSB is 0.75 M t when simulations including uncertainties are examined (compared to 0.5 Mt without uncertainty).

Fishing does not occur every year and the fishery can be closed several years in a row. Landings increased since 2007 from 4 kt to 360 kt in 2011 and 296 kt in 2012 which implies that the exploitation rate (C/SSB) never rose above 10% (ICES 2011a). The spawning biomass was estimated at 2 Mt in 2012 higher than the B_{limit} but more importantly, higher than the target level (0.5 Mt). Under the recommended catch levels, the population is predicted to decrease to a level close to 0.5 Mt (ICES 2011a).

This "precautionary" approach is considered dangerous by some researchers (Ushakov and Prozorkevich 2002). The authors argue that catches were often too large in years 1973-1984 and the minimum spawning stock biomass was too low. They recommend that the minimum SSB be kept at 1.1-1.3 Mt (levels commonly observed in the 1970s) to ensure a minimum fecundity and support the natural dynamics of cohort production under various environmental conditions and the restoration of multi-age structure (Ushakov and Prozorkevich 2002). The discussion continues about how to assess what portion of the population to set aside not only for predators' needs (Ushakov and Prozorkevich 2002) but also as fertilizer for the ecosystem at the price of reducing harvest (Gjøsaeter et al. 2012).

Gradual warming of the Barents Sea over the last 20 years has changed the ecosystem structure. Warmer water species: blue whiting, Norwegian euphausiid species (*M. norvegica*), shrimp have become more abundant in the Barents Sea, and the productivity has increased (Johannesen et al. 2012). Hence, during the 2000s capelin collapse, the abundance of pelagic species (blue whiting, arctic cod and juvenile herring) remained high, and cod biomass continued to increase. The weaker trophic link between capelin and cod is suspected to be caused by the increased productivity and diversity (Johannesen et al. 2012).

This is one of few ecosystems where fisheries management explicitly account for predators' requirements. In addition, predator-prey relationships have also been included in other species' assessment in the Barents Sea (Gjøsaeter et al. 2012).

NORTH SEA SANDEEL

Sandeel, mostly *Ammodytes marinus*, is small, short-lived (~4 years), lipid rich, swims in large shoals and buries itself in sediments in winter. The species is dependent on specific habitat: shallow turbulent sandy area with low concentration of silt and clay. Thus, the presence of sandeel is highly patchy in the North Sea. In addition, eggs are demersal while larvae are only pelagic for 50-90 days (ICES 2010b)[34]. Depending on the currents there could be little connection across the North Sea although modelling suggests connection at the scale of 50-300 km. Settled fish show very limited movements (ICES 2010b), thus, 7 management divisions were defined and within each local differences in productivity recorded (European Commission 2010).

Results from a multispecies virtual population analysis (MSVPA) model for the North Sea indicate that the largest predation of sandeel is due to large fish such as cod, haddock, whiting, herring and mackerel (ICES 2007b). Although birds and mammals are not consuming a large amount of sandeel, seabirds' breeding success depends on the abundance of this high-quality food. It has been noted that several large fish predators (cod, haddock, whiting) show better condition factors in years they have more sandeel in their diet (ICES 2007b). Thus, fishing may have a high risk of local depletion of sub-stocks and have possible impact on predator populations. Sandeel recruitment is strongly affected by plankton production and their timing.

Previous assessments were deemed unreliable but the ICES workshops on sandeel (WKSAN, see ICES 2010b) developed an analytical model (SMS-effort) structured in 2 seasons that uses total catch-at-age, corrected for technological creep to estimate F in a model including year, and age effects, research survey CPUE and also, a stock-recruit function. The survey carried out with a modified scallop dredge since 2004 samples the population in Nov-Dec, when the 0-group has settled. An acoustic survey was developed in the 1990s and data are available since 2007. The natural mortality is the sum of predation mortality at age computed with the MSVPA and 0.2 for other mortality. M has increased in the late 1990s but the assessment uses averaged values over 1982-2000 due to uncertainties and the absence of spatial structure.

The catch potential is largely dependent on the recruitment (Casey and Dôrner 2011). Thus, the stock is also managed with in-season monitoring using the commercial CPUE at the beginning of the fishing season (May) as an index of abundance. The relationship between the early-season CPUE and analytical assessment output is not significant in all regions.

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[34] This section is mainly based on the assessment document (ICES 2010b) unless otherwise noted

For a short-lived species, where an annual forecast is not possible, ICES interprets the MSY concept by using B_{pa} (precautionary biomass) as the escapement, the minimum amount of biomass that must be present before fishing can take place. This is supposed to ensure enough biomass for reproduction and provide sufficient resource for predators[35].

As there are no clear stock recruit relationships, the reference points are defined using historical observations of SSB. B_{lim} is defined as the median SSB in the years of lowest SSB (2000-2006) and no impaired recruitment (ICES 2010b). B_{pa} is defined as a function of B_{lim}[36]. The TAC advice is based on keeping the biomass of sandeel above the B_{pa} level, and in most areas, this limit level turns into a target. In the Dogger Bank area for instance, the expected large recruitment for 2011 led to advice for a catch of a maximum of 320 kt according to the MSY approach because the projected biomass would still be over B_{pa} (200 kt). This in spite of the fact that the resulting F (0.7) would be higher than 2010 (0.4) and twice as much as the 2010 F (0.34) (ICES 2011b). Such a high F was deemed not recommendable. In the last 6 years, quotas have been set at high levels for the whole North Sea in spite of the low biomass (below B_{limit}). Furthermore, management and advice did not follow the harvest control rule (ICES 2011b).

ICES recommended that a management plan should include an upper limit on effort estimated on the basis of the effort applied in the most recent years, and that a target F be defined based on historical data (ICES 2011b).

There is one area closed to fishing in the Northeast UK to maintain high abundance of sandeel for a variety of predators. The area was closed based on evidence of impact of low abundance on kittiwake – the best indicator at this point. The closure has been decreed by the EU for 3 years at a time since 2000 under the condition of providing reports about the effectiveness of the closure on stock rebuilding and the status of kittiwake. Positive results were reported, but it is difficult to differentiate between the effect of the closure and the effects of favourable environmental conditions (Daunt et al. 2008). Re-opening is still possible. MPAs that could be designated in larger numbers in regard to the habitat and bird directive under Natura 2000 raise some questions about future access to sandeel fishing grounds.

NORTH SEA NORWAY POUT

Norway pout (*Trisopterus esmarkii*) is a small gadoid that rarely lives beyond 5 years and most individuals are mature at 2. Thus the fishery is dependent on highly variable recruitment and variation in predation mortality (ICES 2012g, annex 1)[37]. A recent study showed that pouts undergo heavy spawning mortality and natural mortality is correlated with maturation and growth rates (*ibid*). Fishing mortality is relatively low nowadays, so the stock dynamics are more influenced by environmental factors. Still, there is a need to ensure that the stock provides adequate resources for large fish predators that depend on them (e.g. saithe, haddock, cod and mackerel).

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[35] It is noted though that dredge surveys are only available and reliable for 4 of the 7 management areas. Catches in the remaining areas have been low or absent and it was recommended that effort should not be increased (European Commission 2010).

[36] $Bpa=Blim*\exp(\sigma*1.645)$ with $\sigma=0.18$ estimated from assessment uncertainty (area 1)

[37] This section is based mainly on the Benchmark Assessment report (ICES 2012g) unless otherwise noted.

Stock assessment is based on standardised commercial fishing effort and data from the International Bottom Trawl Survey (IBTS). Natural mortality (=0.4 per quarter of a year) and maturity schedule were assumed to be constant over the assessed period. The Benchmark assessment evaluated several models using various scenarios pertaining to variation in M (the most important factor) and maturity, and mean weight at age. M has been estimated using the catch at age from the survey in years of low or absent catches, and with a seasonal multi-species assessment[38]. The two methods predict different patterns at age and, the multi-species assessment results did not lead to a single species assessment that converged. The use of age-specific M, modifies the estimation of long-term recruitment, and leads to a slightly larger biomass (ICES 2012g).

In principle, mortality and catch are set at a level that will maintain the spawning stock biomass over a threshold B_{pa} (150 kt, used as an escapement level and a proxy of B_{MSY})[39] with a high probability of achieving the goal for the next year. B_{lim} is defined as B_{loss}, or the lowest observed SSB observed in the 1980s (90 kt). No reference points for F have been defined. In addition to TACs, the fishery is subject to effort limits, seasons, and reduction of fishing capacity by reducing licences. The Benchmark assessment group suggested increasing the escapement level given the importance of the species in the ecosystem (ICES 2012g).

In 2011, catches were low and F was estimated at 0.034, much lower than the historical average of 0.6 (ICES 2012d)[40]. Evaluation of management strategies concluded that all strategies (fixed F =0.35), fixed TAC (50 kt), and variable TAC escapement strategy) were potentially capable of maintaining biomass above the precautionary level providing that these measures were complemented with additional restrictions. For instance, a fixed quota of 25-50 kt would be successful as long as F remained below 0.6 (ICES 2012i).

Following the MSY approach devised by ICES (ICES 2012d), the current harvest control rule is an escapement strategy deemed well adapted to forage species, utilising information from the first portion of the year and real-time monitoring of the stock since 2006. Yet, the advice for 2012 is a quota that would not exceed 101 kt (F=0.67) which would result in SSB of 150 kt (=B_{pa}), and a 2013 quota <393 kt (F= 1.7) based on projection of the SSB at the end of 2012 and average recruitment. Fortunately, the advice for 2013 will be revised based on in-season estimates of biomass.

Several technical measures have been adopted to protect predators and other species (e.g *Nephrops norvegicus*) from the small-mesh gear used in this fishery: closed Norway pout box, bycatch regulation, minimum mesh size and landing size. Selective grids are under evaluation. The Norway pout box is an area where small-meshed trawls have been banned since 1977 to reduce bycatch of juvenile roundfish (cod, haddock), especially haddock and whiting. It is not possible to fully quantify the benefits to the fishery as no study allows relating the effects of changes in selectivity with changes in catch rates of bycatch species. The box may contain around 30% of pouts numbers based on IBTS estimates in the first and third quarters. Since 2002, the Patch Bank (Norwegian EEZ) has been closed to the sandeel, Norway pout and blue whiting fisheries to avoid illegal catches of juveniles of other species and the Egersund Bank is closed for the winter (December to May) since 2005. Other areas are under evaluation for permanent closure.

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[38] The model is the SMS: a Stochastic Multispecies model forward running age-structured model

[39] Bpa=Blim exp(variance*1.65) where variance=0.3

[40] Catches were low in 2009-2010 because of regulations regarding directed fisheries and bycatch

NORTH SEA SPRAT

The North Sea sprat is a good example of a data poor forage fish. The catch statistics are considered unreliable before 1996. The sprat fishery involves by-catch of herring limited to 10-20% (ICES 2012a). Sprat is short-lived (5 years old are rare in the fishery) and catches are dominated by young fish (age 1-2). Natural mortality is estimated at 0.22 to 0.67 per quarter for ages 1 and 2 for the period 1991-2007 (Total M of 1.77 for age 1 and 1.24 for age 2). The species abundance is monitored using the bottom trawl survey and the acoustic survey. The abundance of 1 year-olds in the trawl survey is used as an index of recruitment.

Only an exploratory age-structured assessment has been presented based on an acoustic survey, trawl survey for quarters 1 and 3, and catch sampling. F for ages 1 and 2 was estimated at 0.49 in 2011 with an average of 0.53 since 2005. No harvest control rule has been derived yet (ICES 2012a). The surveys and the preliminary assessment suggest that biomass has increased since 2000. The quota is set in an ad hoc manner based on the catch of the previous year and the bottom trawl surveys. The ICES advisory committee recommended that the catch for 2012 should not increase from the 2011 catch (134 kt) (ICES 2012e) but the quota, set at 170 kt for 2011 was kept in 2012 (ICES 2012a).

Many predators in the North Sea feed extensively on sprat, including predatory fish, marine mammals and seabirds (ICES 2012a, annex 9). Its role in the ecosystem has been evaluated in the 1981 and 1991 stomach sampling programs. Predation was strongest from whiting and mackerel while predation from cod on sprat seem to increase after the last sampling campaign in 1991, as sandeel and Norway pout stocks decreased. Estimates from 1985 showed that the total seabird consumption of sprat in the North Sea could be as high as removals by fisheries. In winter, sprat becomes part of birds' diet when they cannot obtain sandeels, buried in the sand. However, it is uncertain whether sprat abundance in the North Sea will affect seabird breeding success or overwinter survival. Despite previous attempts to include sprat in the MSVPA for the North Sea, the lack of a single species assessment led to sprat being part of "other food" in the model (ICES 2012a, annex 9).

ATLANTIC MENHADEN

Menhaden, *Brevoortia tyrannus*, can live up to 10-12 years old[41] and reaches maturity around age 2 (ASMFC 2010)[42]. Their core distribution is in Chesapeake Bay, but they migrate along the whole Atlantic coast of the US from to Florida to Maine and juveniles are linked to estuarine habitats. They constitute an important forage fish in the ecosystem.

Although catch has been reported since 1880 its age structure is only available since 1955 which marks the start of the assessment. The fishery is dominated by a reduction purse-seine fishery but menhaden is also landed by the bait and recreational fisheries (ASMFC 2010). The analytical assessment is based on the usual catch sampling data and abundance indices: the age-0 time series from a seine survey, and a pound-net CPUE index series for adults (age 1-3). The seine survey is meant to assess the recruitment strength of other fish while menhaden constitutes only a by-catch. Nevertheless, this is used as the best index available. The pound-nets catch fish mainly from the centre of the population distribution and the CPUE is used as an index of abundance for age 2, and it is correlated with the young of the year index.

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[41] http://chesapeakebay.noaa.gov/fish-facts/menhaden

[42] Unless otherwise noted, the information in this section is mainly taken from the reviewed 2010 stock assessment (ASMFC 2010)

The stock is assessed with the Beaufort Assessment Model, a forward-projecting statistical age-structured model[43]. The assessments used a fixed M=0.45 until 2003 when age and time varying estimates of M (from MSVPA-X, Garrison et al. 2010) were favoured. Estimations of M-at-age vary among methods but the results are consistent with estimates at ~ 1.1 at age 0, 0.8 at age 1, to ~ 0.5 at age 4-10 (ASMFC 2010, p.38). Analysis of age composition of the catch shows a truncation in structure from 1-8 years in 1950-1960 to less than 6 years since 1965.

F_{MSY} and B_{MSY} were not used because the methods to determine these reference points utilise spawner-recruit relationships that are not well defined for the species. Indeed, the relationship between fecundity and age 0 abundance is weak as some of the highest recruitment happened in years of low reproductive capacity. Environmental conditions seem to be an important factor driving recruitment, although the relationship is not fully understood.

Reference points are defined based on a yield-per-recruit model with the intention of protecting the spawning potential and limiting fishing mortality. Population fecundity (number of ripe eggs) tracks reproductive capacity over the years. F_{limit} and F_{target} are set at values corresponding to 15% and 30% of the maximum spawning potential (fecundity per recruit for the unfished stock) respectively (ASMFC 2012)[44]. When F> target of F limit: steps must be taken to decrease exploitation (ASMFC 2011). The stock is overfished as fishing mortality was estimated at 2.28 in 2011 while F_{target} and F_{limit} were estimated at respectively 0.62 and 1.32 (ASMFC 2012). The stock seems to be in a period of low or declining recruitment since the 1990s.

Several indices have been used along the coast to try to understand the extent of influence of environmental conditions on recruitment, mainly related to regional conditions in tributaries. Since 1989, there has been a significant relationship between the young of the year recruitment and annual levels of primary production in Chesapeake Bay, associated with phosphorus loading in Maryland tributaries.

There are fears that the intense concentrated fishing in Chesapeake Bay may cause localised depletion (Menhaden Species Team undated). This would prevent menhaden from fulfilling its role of nutrient recycling and food of numerous predators, including striped bass and bluefish. Other factors that might impact recruitment, such as putative changes in water quality, were not considered formally. More spatially-explicit information is deemed essential (Menhaden Species Team undated). It is the intention of the Atlantic States Marine Fisheries Commission (ASMFC) to move towards a multi-species approach for Atlantic menhaden reference points in the future (ASMFC 2011).

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[43] The model elaborates on CAGEAN and stock-synthesis models. Contrarily to a VPA, the BAM model incorporates error in the catch-at-age and is able to estimate uncertainty in model output.

[44] These reference points are new as the previous ones raised concern about the mismatch between the F and biomass reference points. The next assessment will feature the new developed biomass reference points that will likely show that the stock is overfished.

MANAGEMENT OF FORAGE FISH IN CANADA

Fisheries and Oceans Canada is responsible for the management of all commercial marine and diadromous finfish and invertebrate fisheries in Canada. The management of forage species within Canada is broad, diverse and varies from coast to coast and even stock to stock depending upon the data available, assessment method and the management structure. Very little consideration is given to non-commercially exploited forage species such as Arctic cod and sand lance, although in recent years there has been increased interest in krill as forage for marine mammals. Most transient or migratory species when exploited are managed by effort controls and fishing seasons. Below we describe the general approach of Fisheries and Oceans Canada to the management of commercial species, focus on an ecosystem approach to management and recent policy developments for forage fish, then review the science advice and management of several of the major commercial forage species on both the Atlantic and Pacific coasts.

TOWARDS AN ECOSYSTEM-BASED APPROACH

In Canada the ecosystem approach to management and the incorporation of forage considerations into stock evaluations has evolved slowly. DFO policy toward species interactions and ecosystem structure transformed around 2001 with the Dunsmuir Conference and an initial focus on Large Ocean Management Areas (LOMA). Dunsmuir I (Jamieson et al. 2001) and subsequently Dunsmuir II (DFO 2007c; e), established the process for the ecosystem approach to management, initially as a top down approach, but given several complexities redirected to a bottom up approach under Dunsmuir II. The framework for ecosystem and assessment approach however remained unchanged from Dunsmuir I, although regionally, there have been advances (eg., the DFO Maritimes Region Ecosystem Approaches to Management Framework, see Appendix 1 in Curran et al. 2012). There is currently no national policy for an ecosystem approach to management in Canada, although the need for a National policy was adopted in principle by DFO's Sustainable Aquatic Ecosystem Strategic Outcomes Committee, in June of 2011 (Curran et al. 2012).

In the Maritimes Region the ecosystem approach to management is being advanced through the Ecosystem Approach to Management Framework (EAM) (Gavaris 2009; Curran et al. 2012). Integrated Fisheries Management Plans (IFMPs) are in the process of being developed and implemented to provide a basis for conservation and sustainable use of all commercially exploited species (DFO 2003; O'Boyle and Jamieson 2006). To date the IFMPs have focused on how the resources are being managed with little focus on how to address the data gaps, although in the Maritimes Region reference points have been developed for most major commercial species (Clark et al. 2012). Nonetheless there are real gaps for many of the less common resources.

Currently in Canada, except for new fisheries, commercially exploited forage species are awarded no special status within the overall management regime. The assessment and management practices, although variable from species to species and from stock to stock, are consistent with the approach adopted for harvested fishes. For most species it is simply assumed that a natural mortality of 0.2 or another assigned value is sufficient to meet the required needs of the dependent species.

In Canada key forage fishes tend to be predominantly pelagic species, and because of their abundance, are also subject to exploitation by commercial fisheries. Species such as herring, sardine, capelin, mackerel and shrimp support large and historical fisheries which must compete with fish, marine mammals and seabirds for the limited, and in some cases dwindling, forage base. Other commercial and non-commercially exploited species, such as sand lance, krill, artic cod and anadramous species, also make a significant contribution to the ecosystem as forage.

Category 1. Commercially exploited forage species. These species typically are forage throughout their life history and are very important to the ecosystem. Key species include herring, sardine, mackerel, capelin, and shrimp. In addition to their importance as a forage species, all support significant commercial fisheries throughout their distribution.

Category 2. Non-commercially exploited forage species. Species under this category are important to the ecosystem but are not subject to high levels of fishing mortality. These species are generally not fished commercially, but may be taken as by-catch in other fisheries. Forage species such as sand lance, Arctic cod, and krill fall into this category.

Category 3. Transient and migratory forage species. Typically these species are marine forage during some stage of their life history. Species in this category may be subject to predation during a short time period due to their size or may provide forage during a particular phase of their migration. Many diadromous species enter the marine environment in large numbers as juveniles and are the main forage for a wide variety of fishes and mammals, but are often overlooked in the overall ecosystem scheme.

Under Canada's Ocean Act, the entire ecosystem and the requirements of all stakeholders must be considered in the management of the marine resource and the environment. New initiatives and expansion of the objectives under the Ecosystem Approach to Management require explicit consideration of "forage" within the pelagic habitat and its importance to sustainability within the system. DFO's "Policy on New Fisheries for Forage Species" (DFO 2010c) provides a framework for developing fisheries on forage species. In essence, managers must give due consideration to the impact of a fishery not only on the target species, but the conservation of the ecosystem as a whole. Interactions such as those associated with by-catch need to be documented, evaluated and controlled so that the harvesting of one species is not detrimental to another. An important feature of the policy is the minimization of the risk of change through management targets and the use of the Precautionary Approach to maintain ecological relationships within the bounds of natural fluctuations. The policy concentrates on the management of new forage fisheries and assumes that those planning to exploit the resource will collect the relevant information (DFO 2008; 2010c). For existing commercial fisheries there is no such policy, although consideration is being given to evaluating all commercial forage fish species in the context of the new fisheries for forage species policy.

The Department of Fisheries and Ocean's policy on "New Emerging Fisheries" (DFO 2008) is another policy that has a direct impact on new forage fisheries. Specifically the policy applies to new fisheries for species or stocks that are under or not fully utilized as well as those that do not fall under an existing management plan. The guiding principles include a precautionary approach, evaluation of impact or interaction within an ecosystem context, and a commitment to the collection of appropriate scientific data. Under the Department's vision of "healthy and abundant fishery resources supporting sustainable uses" proponents will be responsible for the provision of reasonable scientific basis to manage the resource from an early stage in the fishery's evolution to the development of precautionary management strategies for harvesting any new or under-utilized species. Unfortunately, like the "Policy on New Fisheries for Forage" it is for new or emerging fisheries. No provision is made to deal with existing commercial fisheries.

Canada's commitment to the implementation of the Precautionary Approach for straddling, and indirectly domestic, fish stocks came into effect in 2001 with the signing of the United Nations Agreement on Straddling and Highly Migratory Fish Stocks (UNFA). In March of 2003 the Government of Canada developed A Fishery Decision-Making Framework Incorporating the Precautionary Approach for all Federal departments. This was followed by a recommendation from the Atlantic Fisheries Policy Review (AFPR) (DFO 2004) to develop a risk management decision framework that incorporates the PA and defined reference points linking stock and ecosystem indicators, objectives for resource and fishery outcomes, and strategies to deal with unwanted outcomes. The decision framework provides definitions and guidelines for the implementation of the framework under a variety of scenarios. Components of the framework include reference points and stock status zones as well as harvest strategies and decision rules for the various stock status zones. Consideration should also be given to the uncertainty and risk associated in reference point and decision rule development.

Unfortunately, many of the key and important forage species do not have defined reference points or reasonable estimates of stock status due to limited data and variable assessment approaches. Furthermore, while the framework provides the foundation for implementation of the PA and what to do under a variety of scenarios, it does not take into consideration the importance or linkages of particular species as forage in the ecosystem. All species are treated equally in the establishment of specific reference points.

HERRING

Herring support important commercial fisheries along the Atlantic and Pacific coasts of Canada (Figure 5) and are one of the key forage species in most regions. It is also a prime example of the diverse and varied assessment and management approaches used in Canada for forage fisheries. Depending upon how the management areas are defined there are 5 main (+2 minor) stocks on the west coast and 10 stocks on the east coast (DFO 2011c; b; 2012c; b; a; 2013). Many of the stocks are assessed and managed differently, although there is some grouping of methods and approaches within jurisdictions.

On the east coast, combined landings of stock complexes for the past two decades ranged between 125,000 and 200,000 t annually with the largest contribution from NAFO Division 4WX (Figure 6). On the west coast total landings in the last decade have decreased dramatically and ranged from 7,200 -12,700 t compared to historical levels of 150,000 - 200,000 t (Figure 7).

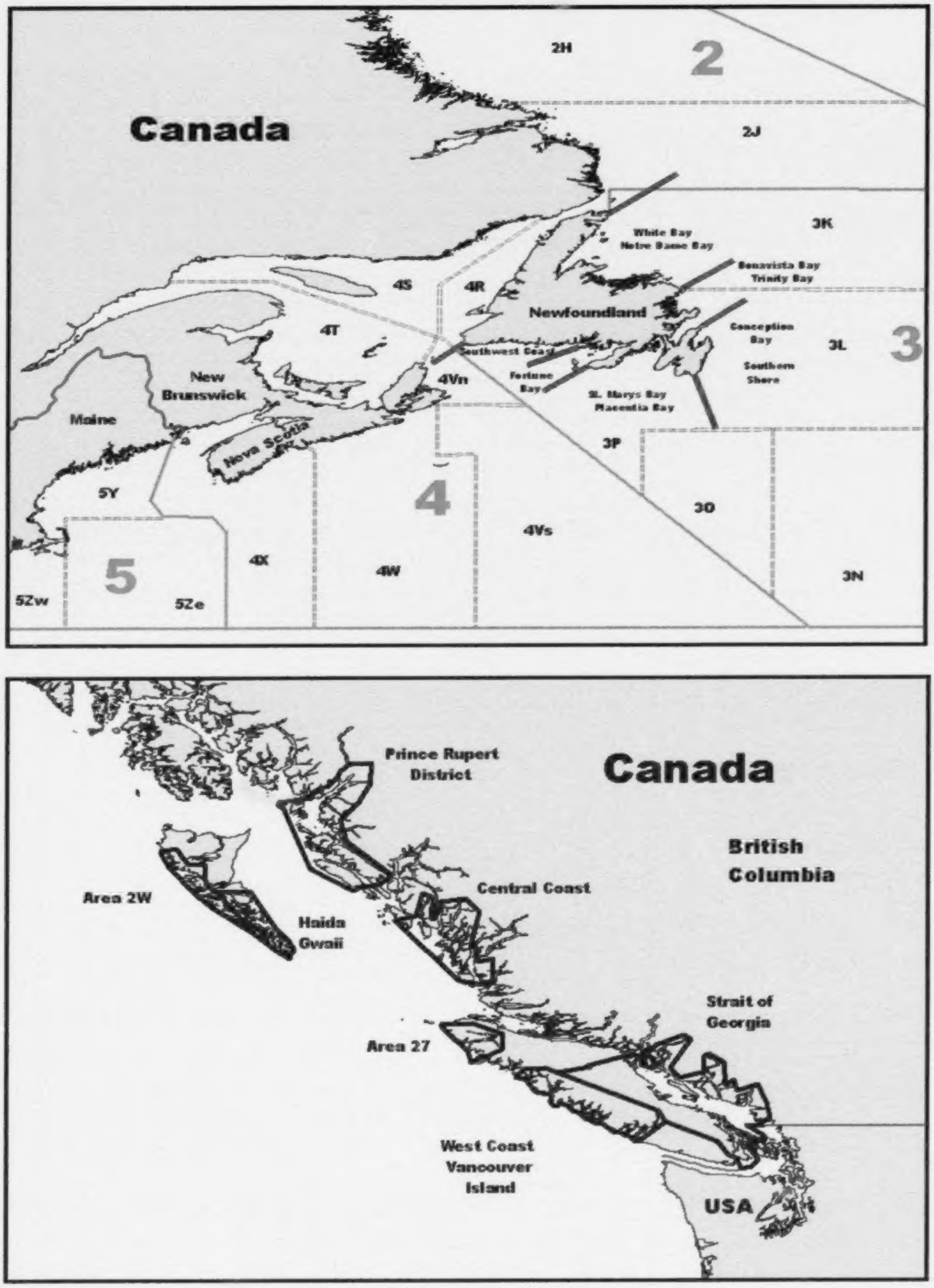
Atlantic herring

Bay of Fundy/ Southwest Nova Scotia

Atlantic herring (*Clupea harengus*) in 4WX were previously (1980-1998) assessed using an age-based analytical assessment (ADAPT) model calibrated with a larval abundance index. The standard target fishing mortality of $F_{0.1}$ was used for management advice, although it was never met. In 1998 the assessment model and abundance index were rejected and the science advice to management was based on 20% (~ $F_{0.1}$) of the observed abundance from acoustic surveys on major spawning grounds (Claytor 2011).

This was further augmented with the implementation of an Integrated Fisheries Management Plan (IFMP) in 2003 which set out the principles, conditions and management measures for 4WX herring including objectives for "the conservation of the herring resources and the preservation of all of its spawning components" (DFO 2003) The main conservation objectives were: to maintain the reproductive capacity of herring in each management unit; to prevent growth overfishing; and to maintain ecosystem integrity/ecological relationships.

Each objective contains several sub-objectives to be considered for evaluating stock status. Recent framework assessments rejected the concept of absolute biomass from acoustics and found unexplainable inconsistencies among several analytical models (O'Boyle 2007; Claytor 2011). At present the 4WX herring stock advice to management is based on trends in acoustic biomass and the limit reference point (LRP) determined from the average observed acoustic biomass between 2005 and 2010 on two of the major spawning grounds. Consideration is also given to the conservation objectives identified in the IFMP in the management of this stock. No assumptions are made regarding natural mortality as only trends in abundance are monitored. The stock moved above the LRP in 2010 and is currently trending upward, although the increases are mainly associated with a single spawning component (DFO 2007b; 2011c; Clark et al. 2012; DFO 2013).


2H
2
Canada
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3K
White Bay
Notre Dame Bay
Bonavista Bay
Trinity Bay
4S
4R
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4T
Conception
Bay
3L
3
Southwest Coast
Southern
Shore
New
Brunswick
4Vn
Fortune
Bay
St. Marys Bay
Placentia Bay
Maine
Nova Scotia
3P
5Y
4
3O
4Vs
4X
5
4W
3N
5Zw
5Ze
Prince Rupert
District
Canada
British
Columbia
Central Coast
Area 2W
Haida
Gwaii
Strait of
Georgia
Area 27
West Coast
Vancouver
Island
USA

Figure 5. Maps of the Atlantic (top) and Pacific (bottom) coasts of Canada.

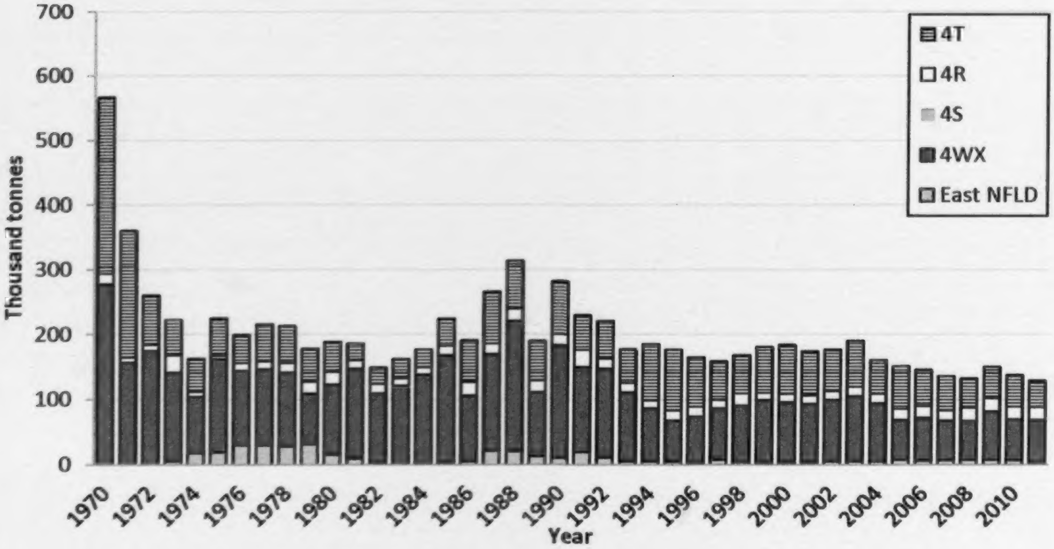

Figure 6. Landings of herring on the East coast of Canada.

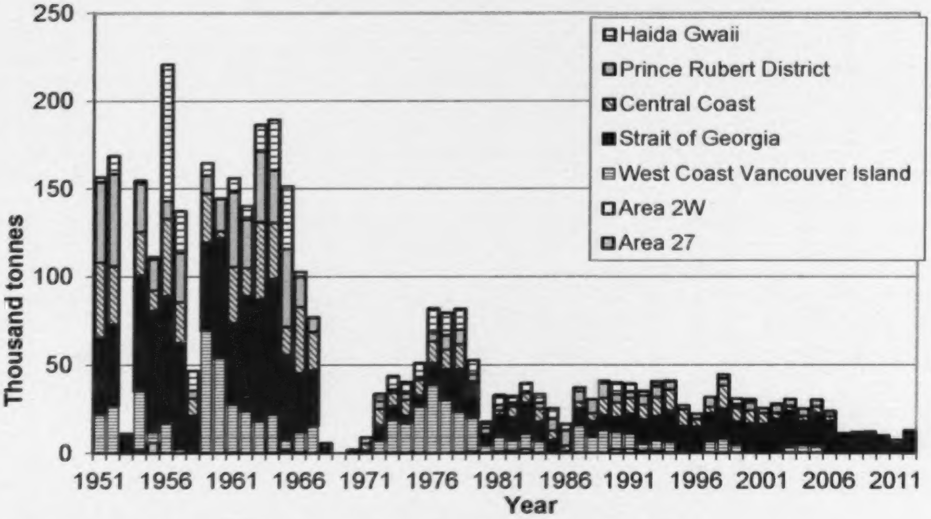

Figure 7. Landings of herring on the Pacific coast of Canada.

Gulf of St. Lawrence

The 4T herring stock in the Gulf of St Lawrence has been assessed with an ADAPT model formulation using several tuning indices and managed with a target reference point of $F_{0.1}$ for several decades. The stock is comprised of many sub-components of spring and autumn (Fall) spawners scattered throughout the southern Gulf of St Lawrence. Each spawning component (i.e., spring and fall spawners) is assessed separately with different target exploitation rates, stock trajectories and limit reference points (DFO 2012b). It is the only herring stock on the east coast of Canada with both LRP and URP (Upper Reference Point) defined. Fall spawners B_{lim} is 51 kt and B_{URP} is 172,000 t, with an $F_{0.1}$ exploitation rate of ~ 25%, and spring spawners B_{lim} is 22 kt, B_{URP} is 52 kt, and exploitation rate is ~27%. Natural mortality is assumed to be 0.2 for all age groups in the ADAPT model, with no consideration given the importance or consumption requirements of other species in the ecosystem. Once the quota has been agreed upon through fisheries management and industry consultations, the quota is divided amongst the sub-components based on historical proportions identified in the IFMP. In 2011, the fall spawners were above B_{URP} and the spring spawners slightly below B_{lim}.

Herring along the Quebec North Shore of the Gulf of St Lawrence (4S) are managed by Total Allowable Catch (TAC) with a fleet sharing formula defined in the IFMP. This is a small herring stock with limited information available to assess the stock and make science recommendations for management advice (DFO 2011a). Other forage species (sand lance, mackerel and Arctic cod) are known to occur in the area, likely in greater abundance. Herring quota is defined in an ad hoc manner based on historical catches and markets.

Newfoundland

Herring on the west coast of Newfoundland and the area south of Cape Charles in Labrador (4R) comprises two distinct stock components: spring and fall spawners. Advice is based primarily on acoustic survey results which have been intermittent over the last 20 years. Prior to 2002 an analytical assessment (VPA – ADAPT) was carried out using the acoustic biomass as a tuning index to provide advice to management. Recently only trends in abundance from the acoustic survey have been used to monitor the stock. The fishery is managed on the basis of a combined TAC for both stock components. The current TAC was set about a decade ago and is based on the last analytical assessment. Specific management measures are in place to protect the spring spawning component. The "evergreen"[45] Integrated Fisheries Management Plan (IFMP) incorporates management measures for fixed gear fisheries and fleet quota sharing (DFO 2010a).

The south and east coast Newfoundland herring fishery comprises 5 independent stocks and the majority contain spring and fall spawning herring. Major changes in the proportions of spring and fall spawning herring have been observed in recent years, which may be the result of generally increasing water temperatures or climate change (Melvin et al. 2009; Wheeler et al. 2010). Historically, analytical assessments were performed for each of these stocks to provide advice to management. However, since 2002 scientific advice has been based on performance indicators through evaluation of abundance indices and biological characteristics. The stocks are managed by TAC and IFMPs exist for all stocks (DFO 2010a). Most of these fisheries are limited by available market. In recent years TAC's have not been restrictive and are based on historical levels.

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[45] evergreen was used to say that there is no termination date.

Pacific herring

The Pacific herring, *Clupea pallasii*, fishery operates over a broad geographical range and is divided amongst 5 major and 2 minor stocks: Haida Gwaii (Area 2E), Prince Rupert District, Central Coast, Strait of Georgia, and West Coast of Vancouver Island, with minor herring stocks in Area 2W and Area 27 (Figure 5). These stocks have a long history of analytical assessments, utilizing a variety of models and abundance indices, upon which advice has been based. The current model is an integrated statistical catch age model (ISCAM) (Cleary and Schweigert 2011). The fisheries are managed under an IFMP and a Management Framework to establish TAC's for each stock (DFO 2011d). Harvest control rules set the maximum available commercial harvest for each of the major stock areas at 20% of the forecast of mature stock biomass (males and females combined) when the forecast of mature stock biomass is above the commercial fishery threshold or "cutoff", which is 25% of the estimated unfished biomass (0.25 B_0). If a forecast exceeds a cutoff, but a 20% harvest rate would result in spawning biomass that is less than the cutoff, the maximum available harvest is determined as the difference between the forecast and cutoff. Three of the 5 stocks have not had a commercial fishery for 5 years or more and only a "spawn on kelp" fishery has occurred in the 2 minor stocks (Figure 7).

MACKEREL

Atlantic mackerel, *Scomber scombrus*, migrates between southern latitudes in the winter to northern habitats to spawn as far north as the southern Gulf of St. Lawrence (Studholme et al. 1999). Its distribution strongly responds to changes in water temperature with preferred temperature ranging from 9 to 13°C (Overholtz et al. 2011). In the last 40 years (1968 and 2008) both annual variability and decadal trends of climate-driven changes in the marine environment resulted in changes in spring distribution area and mean latitude, moving from the mid-Atlantic region to the New England and even the Grand Banks (Overholtz et al. 2011). Migration patterns have also changed in Canadian waters (Grégoire et al. 2010). During this period, landings have declined in the mid-Atlantic coast and southern New England while they increased in Canada. Mackerel is assessed jointly by DFO and NOAA by a Transboundary Resource Assessment Committee (TRAC) (Deroba et al. 2010).

The 2009 assessment, based on a VPA model, was found unsatisfactory as all models led to strong bias and retrospective problems that shed doubt on computed reference points (Deroba et al. 2010)[46]. Since then, the bottom trawl survey, which was used to calibrate the stock assessment, was rejected as a suitable index of abundance of the northern contingent (Grégoire et al. 2010). Thereafter, two different types of analytical assessment were performed. First, traditional cohort analyses (VPA) were used, without a time series to calibrate the VPA and using different starting parameters, to estimate trends in abundance, fishing mortality and reference points (Grégoire and Maguire 2010b). Second, several versions of an XSA model (Extended Survivors Analysis) were utilized to evaluate stock status (Grégoire and Maguire 2010a). Although these trials were not found satisfactory and the output values differ, they produced similar trends in biomass and fishing mortality. These results also suggest that the data may be incomplete, for instance, bait and recreational landings are not reported in Canada (Grégoire and Maguire 2010a).

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[46] the model is faced with contradictory trends between catches, spring survey and commercial CPUE indices

More recently, an analytical assessment was carried out with a sequential population analysis using the total catch-at-age (1968-2011) and the spawning biomass index from the Southern Gulf (1996-2011) (DFO 2012e). The spawning biomass index (based on daily fecundity and total egg production methods) shows that the biomass in the southern Gulf has been at a historical low since 2005[47]. Estimated fishing mortality rates increased in the 1990s reaching over 0.5 in 2008. The 2010 assessment concluded that F should be reduced to 0.124 on the basis that this was the average for 1968-1992 period (a sustainable catch period) with a catch set at ~8700 t in 2012 and 2013 (DFO 2012e).

Canadian landings have been above average for 3 of the last 5 years while US landings have declined substantially. There has also been a major shift in distribution since 1995 and significant increases in landings off the east coast of Newfoundland beginning around 2000. Canadian catches increased between 2001 and 2006 because of the recruitment of a strong cohort (1999) and increase in effort (DFO 2012e). Since 2006, they decreased and reached their lowest values in 2011. TACs were decreased to 80,000 t in 2010 and 60,000 t in 2011, a threshold that did not constrain the catch of less than 10,000 t for both US and Canada (DFO 2012e).

CAPELIN

Capelin (*Mallotus villosus*) is a small pelagic species common throughout shelf waters of the north Atlantic. In Canada it occurs primarily off Newfoundland and Labrador where it is considered a key forage species. The NAFO SA2, Division 3K and Division 3L stocks have been treated as a single stock complex since 1992 (DFO 2010b). Catches in the offshore peaked in the late 1970s (250,000 t) during the foreign fishery, which closed in 1979. This was followed by a Canadian inshore fishery that reached ~80,000 t between 1988 and 1990, but rapidly declined to around 30,000 t in the early 1990s. Consistent with the decline in landings was a significant decrease in abundance that was associated with a shift in spawning time (June to August) (Mowbray 2013). The shift was attributed to changing environmental conditions and likely responsible for low recruitment. Currently there is no analytical assessment model for the stock. Abundance is based on broad scale acoustic and spawning site (egg deposition, hatching) surveys from a few index sites to evaluate trends in stock abundance. In theory, consideration is given to capelin as a forage species and its importance in the ecosystem. The stock is supposed to be managed based on a conservative exploitation rate of 10% of the projected spawning biomass (implemented in 1979, DFO 2010b), but predicted stock biomass has not been available since 2001. Several Integrated Fisheries Management Plans have been developed for the capelin fishery since the early 2000s (DFO 2010b). The TAC was reduced in 2011 and remains at one of the lowest levels in the history of the fishery. The stock is considered to be at a relatively low level compared with the 1970s (DFO 2012d), although between 2011 and 2012 there were signs of improvement with changes in distribution and timing of spawning moving more toward those observed during peak abundance. Climate is a very important driver of population dynamics and phenology, and lead to variations in accessibility over the years (Nakashima 1996; DFO 2010b).

EULACHON

Eulachons, *Thaleichthys pacificus*, are distributed along the Pacific Coast of North America from the Aleutians in Alaska to northern California (COSEWIC 2011). It is a smelt that spawn in coastal rivers associated with glaciers and snowpacks. They live 3 years in the ocean in coastal waters at depth of 50-200 m, and migrate to freshwater when they reach 3 years old for their unique reproductive event (COSEWIC 2011). They are exceptionally rich in lipid (20% of the wet weight) and as such are of the utmost importance for First Nation communities and constitute an important forage fish for numerous predators (COSEWIC 2011).

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[47] Mackerel biomass also declined on the Scotian shelf and the western coast of Newfoundland over the last decade, owing to a combination of lack of recruitment and higher than sustainable fishing mortality (DFO 2012e).

Their populations[48] have been globally declining in coast-wide synchrony for the last 30-40 years to very low levels, and extirpated in several rivers. Populations from southern Rivers were the most affected for reasons that are not well understood (COSEWIC 2011). The species has been listed as endangered in the United States and in British Columbia. Since 1995 DFO has suspended exploitation in the Fraser River, and several measures pertaining to limited entry in the fishery, habitat protection and bycatch control have been established (Hay et al. 2003; COSEWIC 2011).

Only the Fraser River supported a commercial fishery (and native fishery) while First Nations traditional fisheries occur in 14 other rivers (Hay et al. 2003). There is little known about the population dynamics of these stocks, and there is no stock assessment. The management of the Fraser River population is based on pre-season indicators as reference points: the SSB for the previous two years; the offshore biomass index from the previous year and current year catches in the Columbia River[49]. The New Westminster Eulachon test fishery (gillnet) providing the cumulative catch as an in-season indicator has not been in operation since 2005 (Hay et al. 2003; DFO 2011e).

Caution is recommended when the index falls below a determined threshold on the western coast of Vancouver Island (500 t in area 124 and 1000 t in area 125). Both areas show abundances below the reference levels consistently (DFO 2011e). The management rule for this species is plagued with a lack of basic information and short time series. For this reason, reference levels were chosen ad hoc. A low spawning stock biomass is defined as 150 tonnes and is a cause for caution if it happens one year, a second consecutive year is cause for concern and would result in a full stoppage of removals (Hay et al. 2003). Since 2004, none of the estimates have risen above 50 tonnes (DFO 2011e). Based on these reference points and the low biomass estimated, there was no commercial or recreational fishery in 2011 and First Nations fisheries were subject to negotiations.

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[48] The number of distinct populations is unclear and several types of groupings could be justified based on genetic variations (COSEWIC 2011).

[49] The Columbia River catch is used as an indicator of ocean survival (DFO 2011e) assuming that their return in the Columbia (Jan-March) is an index of the level of return in the Fraser (March-May). Although low catches in the Columbia give a rough indicator that the Fraser return might be low, high catches offer no guarantee for the Fraser (Hay et al. 2003). Columbia River catches below 500 t are used as an indicator to close the Fraser River.

LESSONS FOR CANADIAN FISHERIES

SUMMARY

Evaluation

In the sections above, both Canadian and international case studies informed us of the current state of commercial forage species fisheries and their management. In this section, the information is summarized in three categories: the importance of the forage fish judged from the number of forage fishes in the ecosystem (key, when the species is the most abundant forage species, or important when it is one of many forage species); the basis for management; and the ecosystem considerations included in the fishery management. In addition, the management is evaluated using criteria pertaining to the definition of control rules, its implementation and the current status of the stock. The evaluation is qualitative and meant as a way to draw lessons rather than an absolute score.

A fishery that has a defined harvest control rule with upper and lower reference points is considered here as on a positive path (Table 1). It was beyond the scope of this document to evaluate the reference points but there is a special mention for the use of B_{pa} as the upper reference point because B_{pa} is defined as "B_{lim} + the variance in historical data", to add a buffer accounting for the uncertainty in the minimum biomass (ICES 2007a, Book 1). The second part of the evaluation conveys the success at following the rules, controlling fishing mortality, and the status of the stock. The latter is based on the conclusion reached by each agency and thus, does not account for the biases resulting from deficient reference points. Stock status and the level of fishing mortality are designated as unclear when there is a discrepancy between the stock assessment and the reference points. Finally, there is tendency for agencies to use the limit level as a target level rather than an escapement and a level of biomass to be avoided (Piet and Rice 2004). This was evaluated by looking at the projected biomass given the recommended quota and the rationale for choosing such quotas.

The score for each stock was computed as the number of points earned divided by the maximum score (8). In cases where the response was "n/a", this was subtracted from the maximum score prior standardisation. The results are detailed in Tables 2 and 3.

Table 1.Evaluation criteria for case studies

No.	Criteria	Classification	Points *a*
	Harvest rule definition		
1	Reference points defined	yes, no, obsolete	1,0,0
2	Escapement or min biomass (B_{lim})	yes, no	1,0,
3	Target level defined	yes, no, B_{pa}	1,0,1
	Implementation		
4	Management follows control rule	poor, fair, good	0,1,2
5	F compared to recommended F	lower or equal, higher, unclear	1,0,0
6	Stock status	cause for concern, no concern, unclear	0,1,0
7	Use limit ref. points (B_{lim}) as target	yes, no	0,1
	Maximum total		**8**

a When a criterion does not apply to the case study (n/a), the item is removed from total prior to standardisation

Results

Several stocks described here suffered at least one collapse in the last 40 years. The rebuilding of these stocks was achieved by a severe reduction of the fishery and the occurrence of one or several large cohorts owing to favourable environmental conditions (e.g. all herring stocks, Barents Sea capelin).

The reference points are defined for most European stocks (Table 2) while the work is still incomplete for their Canadian counterparts (Table 3), explaining in part the generally lower scores for the Canadian forage fisheries. This classification hides the fact that most target reference points are very uncertain (e.g. U.S. and Canadian herring) or a simple variant of B_{min} (e.g. ICES stocks). In the latter case especially, it sets low expectations and results in a tendency towards overly optimistic setting of quota and determination of stock status. The 12 stocks (out of 20) for which there was no concern (the biomass would remain over the threshold) were divided in 2 groups defined by the quality of their implementation. The 8 stocks with generally high scores were all stocks of sardines, Barents Sea capelin, herring from Alaska, Norwegian Sea and Gulf of Maine, and the menhaden. They distinguish themselves also by the level of investment in the assessment, auxiliary information and fishery monitoring. Although the menhaden stock is not considered overfished and has a high score, the target F (0.62) seems quite high for a species that sustains so many predators, and the realised fishing mortality reached 2.28 in 2011.

The stocks with lower scores were the ICES stocks, characterized by low reference points and poor adherence to harvest control rules. The status of most Canadian stocks was described as “cause for concern” or “unclear” for various reasons, ranging from adverse environmental conditions (e.g. capelin), confusing signals in the stock assessment (e.g. mackerel) or a combination of both (e.g. possibly herring in SW Nova Scotia). The situation of eulachon, characterized by poor information and possible environmental effects illustrates the situation of most diadromous or transient forage fish in Canadian marine waters.

In most case studies there was effort made towards defining simple harvest control rules or at least reference points. For instance, Alaskan herring relies on in-season surveys and modelling with low exploitation rates meant as precautionary (e.g. Alaskan herring). The South African anchovy and sardine management is quite complex, considers a lot of factors and now includes a possible switch in species dominance. Its major achievement is to have tested the robustness of management rules in the face of data, model and implementation errors. Nonetheless, in such a widely fluctuating ecosystem, the management procedure was progressively improved by adding variables and constraints to account for changes in climate regimes and other possibilities.

Only a few assessments used estimates of natural mortality that include predation mortality. In the case of Gulf of Maine herring, the assessment model including predation mortality became problematic and was abandoned. That of the North Sea herring was kept but unfortunately, the resulting high biomass was still compared to obsolete biomass reference points. Unfortunately, reference points based on yield-per-recruit models lead to unrealistic estimates of reference fishing mortality that increase with M. A complete process would include re-estimating the reference points as suggested by the ICES Advisory committee, but also subtract the biomass consumed from the TAC (as it is done in the Barents Sea). There is also confusion between the need to account for predation in the assessment, the resulting higher biomass and the meaning of reference points to calculate quotas.

Historically, most research and advances have occurred primarily in assessment methodology. Although advances have been made in multispecies models and diet studies, the inclusion of predation mortality in the assessment model has rarely decreased the estimation of TAC. Only in two ecosystems (Barents Sea, Antarctic) were specific allowances made for predation. In the Barents Sea the study of ecological processes and stock interactions proved to be of tremendous help to build multi-species models and complement population dynamics (Gjøsaeter et al. 2012). From these studies it became clear that fisheries and cod were competing for the same fish and, to avoid further collapses, cod predation was subtracted from the calculated TAC. The Antarctic krill fishery has set a precautionary limit and spatial restriction to avoid local depletion that would affect breeding birds (Constable et al. 2000).

Auxiliary information and ecological studies were instrumental in understanding the causes of collapse and errors in some assessment (eg Barents Sea capelin, Norwegian sea herring, North Sea sandeels and Norway pout), and in defining management problems and finding solutions in others (e.g. Benguela, see above). The role of climate, but also zooplankton production and their structure in size and species composition, was instrumental in understanding the ecological mechanisms behind the observations.

Only a few case studies included climate information. Changes in carrying capacity are now included in the harvest rules for South Africa after the sudden change in species dominance and its effect on fisheries management (De Oliveira and Butterworth 2004). Climate indices are included in the TAC calculation for the Eastern Pacific sardine, but even in this case, while the existing (but unused) auxiliary data on impending changes in the climate regime and current changes in population structure indicate that this stock could be on the verge of a collapse, fisheries management follows its course of not fully taking into account factors such as fish migration on catches. In the Northwest Atlantic (Canada and US), despite the increasing indications of the importance of climate in stock production, very little of this information is included in assessments and management advice (Link et al. 2011, and this study).

Ecological studies have been carried out for a long-time in the North Sea and this information has been used to build sophisticated multi-species models. However, this information is still not entirely integrated in forage fish management, except to include better natural mortality estimates in assessments, and to define closed areas to protect species from by-catch or protect seabirds from local depletion, although these are still disputed. The escapement levels set to protect the population and predators are still not grounded in formal knowledge of predators' consumption.

The definition of reference points and harvest control rules does not constitute the end of the management process. Unforeseen natural events such as changes in climate regimes (and ensuing changes in productivity) and in trophodynamics can render these rules obsolete, requiring rapid adaptations. This was obvious in the southern Benguela and could happen in Northern Humboldt (Peru) or the California Current, and highlights the need to be adaptive and to continue looking at the stocks with more than one index and one perspective.

Table 2. Characteristics and evaluation of forage fish management for international case studies.

Species – area	Role of forage species *a*	Basis for management	Ecosystem considerations	Evaluation
Krill – Antarctica	Key	• Limit and target biomass. • Scientific surveys for target species and predators (birds, mammals). • Spatially explicit • No fishery monitoring	• Take into account predators' needs explicitly. • Lower quota than potential, maximal quota set. • Preliminary spatial allocation of catch and one being developed.	1. yes 2. yes 3. n/a 4. good 5. lower 6. no concern 7. no Score=1
Herring-Alaska	Important	• Limit biomass and 10-20% exploitation rate • Egg deposition survey • Spatially explicit • Age-structured or biomass projection assessment	• Need of predator taken into account implicitly, by using a relatively low F. • Spatial structure to avoid loss of aggregations.	1. yes 2. yes 3. n/a 4. good 5. n/a 6. no concern 7. no Score=1
Herring-Norway	Key	• Control rule featuring a lower and upper limit for biomass and varying F as a function of biomass • Rebuilding phase: F=0.05 • Analytical assessment using several types of surveys	• Low F target (0.125) to decrease the risks of overfishing	1. yes 2. yes 3. yes 4. fair 5. higher 6. no concern 7. no Score=0.75
Herring-North Sea	Important	• Limit and trigger ref. points for biomass, and varying target F with biomass • Fishing allowed under Blim • several surveys used for indices • Age-structured analytical assessment • Predation mortality included in assessment	• Predator removal used to calculate M but not accounted for in TAC • Need of predators unknown • Lower F on juveniles	1. obsolete 2. yes 3. Bpa 4. fair 5. unclear 6. unclear 7. yes Score=0.37
Herring-Maine	Key	• MSY–based reference points found very sensitive and uncertain • Several surveys used for indices • Age varying M • Age structured assessment	• Use knowledge of predation to improve assessment but not to modify exploitation rules • Informal account of ecosystem needs	1. yes 2. yes 3. yes 4. good 5. lower 6. no concern 7. no Score=1

Species – area	**Role of forage species *a***	**Basis for management**	**Ecosystem considerations**	**Evaluation**
Sardine anchovy - South Africa	Key	• 2 acoustic surveys • Management procedure with in-season TAC adjustment • Constraints on F and quota at min and max biomass • No analytical assessment	• None in the assessment • Account for bycatch • Account for change climate regimes • Closed areas for seabirds	1. yes 2. yes 3. yes 4. fair 5. n/a 6. no concern 7. n/a Score=0.83
Sardine-Eastern Pacific	Key	• 3 types of survey: egg production estimates, aerial and acoustic • Analytical assessment • Control rules based on escapement biomass and exploitation rate of 5-15% • Geographical distribution of the stock not completely accounted for in catch allocation • Suggestion that collapse may be looming based on auxiliary indicator	• No explicit ecosystem consideration	1. yes 2. yes 3. n/a 4. good 5. lower 6. no concern 7. no Score=1
Anchovy-Peru	Key	• Acoustic surveys per season • Fishery monitoring • Precautionary reference points defined • The limit set on F may be too high • Several measures to control the fishery	• Monitoring of predators • Numerous ecological studies to help understanding the role and trophodynamics of this species • No explicit consideration for predators' consumption	1. yes 2. yes 3. no 4. good 5. n/a 6. no concern 7. yes Score=0.75
Capelin-Barents Sea	Key	• Acoustic survey • Multispecies model used to include predation in assessment • Cod stomach content analysed annually	• Season defined to account for cod feeding before spawning • Cod consumption and level of herring migration in the Barents Sea included directly in the assessment model	1. yes 2. yes 3. yes 4. good 5. n/a 6. no concern 7. no Score=1
Sandeel-North Sea	Important	• Research survey, in-season monitoring using commercial CPUE, dredge surveys • Analytical assessment • Account for predation mortality using MSVPA • Escapement biomass • Need upper limit of effort and target F • Spatially structured	• Some closed areas to protect predators (birds) from localised depletion, being evaluated • Escapement strategy aim to keep enough biomass for predators (but needs well defined?)	1. yes 2. yes 3. Bpa 4. poor 5. n/a 6. no concern 7. yes Score=0.57

Species – area	**Role of forage species *a***	**Basis for management**	**Ecosystem considerations**	**Evaluation**
Norway pout- North Sea	Important	• Trawl survey, standardised commercial effort • In-season monitoring of the stock • Analytical assessment • Current harvest control rule based on escapement and historical data • Closed areas and by-catch regulation to protect other species	• Not possible to evaluate benefits of closed areas • Known dependent predators but did not quantify needs • Closures implemented to limit bycatch	1. yes 2. yes 3. Bpa 4. fair 5. n/a 6. no concern 7. yes Score=0.71
Sprat- North Sea	Important	• Data poor • Acoustic and trawl survey • Analytical assessment in development • No harvest control rules	NA	1. no 2. no 3. no 4. poor 5. n/a 6. no concern 7. n/a Score=0.17
Menhaden- NE USA	Key	• Age 0 seine survey, age 1-3 pound-net index • Analytical assessment with age-dependent M • Reference points based on yield per-recruit to protect reproductive potential • F corresponding to SSB =15-30% of maximum spawning potential (F=1.32 and 0.62)	• Goal is to protect spawning potential and forage function but current F=2.28 >> ref. points • Intention of addressing address multi-species needs in the future	1. yes 2. yes 3. yes 4. good 5. higher 6. no concern 7. no Score=0.87

Table 3. Characteristics and evaluation of forage fish for Canadian case studies.

Species – area	**Role of forage species**	**Management tools**	**Ecosystem considerations**	**Evaluation**
Herring-SW Nova Scotia	key	• Acoustic surveys of major spawning grounds. • Closed areas, Quota restrictions. • Extensive monitoring of fishery	• No special consideration for ecosystem. Standard M=0.2	1. yes 2. yes 3. no 4. poor 5. unclear 6. concern 7. yes Score=0.25
Herring-Gulf	key	• Fishery dependent and independent surveys. • F0.1 target. • Quota, seasons, minimum size. • Age structured analytical assessment for spring and fall spawners. • Extensive monitoring of fishery. • Control rule limit and target reference point for SSB	• No special consideration for ecosystem. Standard M=0.2	1. yes 2. yes 3. yes 4. good 5. lower 6. Fall: no concern, Spring: concern 7. Yes Score= 0.62
Herring-Nfld	important	East: • Multiple stocks. Trend analysis of indexed fishermen CPUE. • Several indices of abundance. • Small market driven fishery. • TAC not prohibitive. • No reference point. • Series of performance indicators with control rules West: • Acoustic survey, analytical assessment. • No analytical assessment. Quota taken or exceeded.	No special consideration for ecosystem requirements.	East: 1. No 2. No 3. no 4. Fair 5. n/a 6. Mixed 7. No Score= 0.28 West: 1. No 2. No 3. no 4. n/a 5. Unclear 6. Concern 7. No Score =0.14
Herring Pacific	key	• Multiple Stocks, analytical assessment, egg bed surveys, reference points and decision rules. • 5 of 7 stocks currently closed to fishing.	No special consideration for being a key forage species or meeting the ecosystem requirements.	1. Yes 2. Yes 3. yes 4. Fair 5. Unclear 6. Concern 7. Yes

Species – area	Role of forage species	Management tools	Ecosystem considerations	Evaluation
		- Constant harvest rate of 20% and minimum escapement strategy $0.25B_0$ (LRP)		Score=0.375
Mackerel-NW Atlantic	important	- Problems with analytical assessment, in re-development - Trawl survey in the Gulf of Maine not representative - Spawning biomass index in the Gulf - Reference points based on historical data only	None	1. undocumented 2. yes 3. no 4. good 5. higher 6. unclear 7. n/a Score=0.5
Capelin	Key	- Single Stock Complex. No analytical assessment. Abundance based on acoustic survey of juveniles. - Decision rules defined, but not applied due to lack of biomass estimate.	Acknowledgement of ecosystem importance, but nothing specific applied.	1. No 2. Yes 3. no 4. Fair 5. na 6. Concern 7. No Score=0.5
Eulachon	Important	- Information partial for most populations - Important influence of climate on their dynamics/decline suspected - In season indices - No assessment		1. ad hoc 2. yes, ad hoc 3. no 4. good 5. n/a 6. concern 7. n/a Score=0.57

LESSONS LEARNED

Reference points and harvest control rules

Most forage species are characterised by large and rapid fluctuations in abundance due to environmental variations, and thus, predators also experience these low abundance cycles that deprive them of important food source. Overexploitation adds to this vulnerability and, although depleted forage fishes have mostly recovered around the world, recovery can take decades and some have not fully recovered in 40 years (e.g. Namibian sardines, Canadian southwest Nova Scotia herring). It is becoming clear that the management objective should be to preserve a sufficient biomass for predators thereby decreasing the probability of collapse. This would protect forage fish stocks, the fisheries directed at forage species and their predators as well as ecosystem structure and functioning.

Harvest control rules are useful management strategies that clarify the course of action to be taken to reach the management objectives. They reduce the annual discussion and bargaining among parties and to have the capacity to forge cooperation between stakeholders (Cochrane et al. 1998; Gjøsaeter et al. 2012). In most case studies considered above, there was an effort to put in place a system to monitor the population and define harvest control rules to restrict catches. Nevertheless, there are still several questions concerning the selection of reference points used in harvest control rules and the way they are implemented.

Managing with reference points is not without uncertainty associated with their estimation. Frequently, the definition of reference points relies on historical trends/averages and other proxies that are potentially misleading, depending on the window of observation (e.g. European waters Guénette and Gascuel 2012). The Canadian guidelines follow this trend (see Appendix 2). It is debatable whether target biomass can be defined from a limit threshold (e.g. ICES where $B_{pa}=B_{lim}$+buffer) and used as a B_{MSY} proxy for short-lived forage fish because B_{MSY} is unlikely to provide a buffer against uncertainty, let alone respond to predators' need. In addition, these low precautionary reference points, especially the limit points, have been increasingly used as targets rather than limits to avoid (Piet and Rice 2004) which had the perverse effect of keeping the stocks at low levels (Guénette and Gascuel 2012). Several of the case studies presented above suffer from this bias.

Most reference points, either model-based or empirical indices, are not absolute quantities that will never change. The factors that are known to influence the estimation of reference points, growth rates, natural mortality, selectivity, stock-recruitment and carrying capacity relationships are subject to error and will change with environmental conditions thus, they should not be considered as stationary (Mohn and Chouinard 2007; ICES 2010a). These factors should not be omitted from modelling exploration or provision of fisheries management advice.

Ecosystem considerations

Ecosystem overfishing can occur when the harvesting of prey species impairs the long-term viability of other ecologically important species (Pikitch et al. 2012a). At this point in time, progress in including ecosystem considerations into assessment and management has been slow due to inertia in fisheries agencies, lack of data, or both (this study and Link et al. 2011). In most assessment documents the chapter on ecosystem consideration consists of an overview of possible fisheries interactions (by-catch), habitat problems and known predators. Only in a few ecosystems, does fisheries management integrate ecological information and/or restricted exploitation to protect predators' foraging needs.

In several ecosystems, predation mortality is estimated to provide more realistic natural mortality estimates in the forage fish stock assessment but it does not go as far as to use this information to limit quotas. The Maine herring recommendation set the quota then subsequently informally verified that the predators' consumption requirements would still be accommodated. This process may not be sufficient if forage biomass declines, prompting the need for a hard decision between quota status quo and predators' need. Finally, in most other assessments reviewed here, no formal inclusion of predation information is included in forage fish management (e.g. Canadian fisheries, California sardines, Peruvian anchovy).

A few examples include climate indices, acknowledging the observed and expected changes in carrying capacity. Several authors called for using both biomass estimates and climate indices (e.g. Jacobson and MacCall 1995 in Boyer et al. 2001) to set quotas but more importantly to test the robustness of management plans. This is a daunting task because of the difficulty to predict future environmental conditions and precise effects on forage fish. Nevertheless, ongoing international projects are attempting to tackle this question for European waters (e.g. FACTS North Sea). In the meantime, climate indices are used more informally in some ecosystems (e.g. Northern Humboldt, Southern Benguela) to define management rules and guard against overfishing.

The noted lack of implementation of ecosystem-based management does not necessarily reflect the lack of interest from scientists but rather the necessity to learn new ways of providing advice, the presence of inertia in fisheries policy and management, a lack of scientific direction and of basic ecological data. From a technical perspective, various ecosystem and multi-species models have been developed to place fisheries in an ecosystem context and evaluate management scenarios in numerous countries (Plagányi 2007). They were built for various objectives ranging from inputs to assessment models, furthering understanding of multi-species interactions, to represent partial or full ecosystems, or to estimate overall system productivity (ICES 2012h; Link and Bundy 2012). They generally include trophodynamics and sometimes climatic or geochemical input, and increasingly, uncertainty (ICES 2012h).

Evolution not revolution has been the mantra in developing an ecosystem approach to management, as illustrated by management of the Baltic Sea, the Barents Sea, and Alaska. Ecosystem considerations will necessarily start with the most obvious issues at hand and use ecological knowledge to guide management and solutions (Casini et al. 2011; Link and Bundy 2012). Perfect knowledge is not required to act and include ecosystem considerations into management, as experiences in Norway, Alaska and the Antarctic have demonstrated. The most difficult part is to convince all parties of the need for conservative management measures that consider other species needs to be precautionary. As more factors are taken into account, managers face conflicts from multiple sources of influence and of the associated trade-offs (Casini et al. 2011).

Decision overfishing

Beyond the difficulty of defining meaningful reference points and harvest control rules, the commitment to respect the harvest control rules and make the hard decisions is instrumental to success but often constitutes the problem. Unfortunately, "decision or regulatory overfishing", the inflation of TAC beyond the scientifically set quota to satisfy short-term social or economic imperatives (Eagle and Thompson Jr 2003; Aps et al. 2007), is wide spread around the world because of a definite reluctance to implement necessary cuts in harvests when needed (Mace 2001).

Numerous case studies illustrate the tendency to choose short-term benefits over the need for precaution and conservation (present study and Cochrane et al. 1998; Boyer et al. 2001; AFH 2012a; b; Brodie et al. 2012). In the Baltic Sea for example, regulatory overfishing could be the major reason depleted stocks have not recovered (Aps et al. 2007; Aps and Lassen 2012). The official TACs have been 53% higher than scientifically-recommended before 2002, and 104% higher since 2002. This is generally the case in European waters where the European Commission management decisions are often opposed by national ministers, resulting in increased quotas more than 60% of the time (O'Leary et al. 2011; AFH 2012b, and cases studies in this report). In Canada and NAFO the scientific advice has frequently been overridden in spite of the official adhesion to the precautionary principle (Shelton 2007; Brodie et al. 2012, and this study).

Given a choice, managers will choose higher quotas than the best estimate or the mid-point fisheries advice, ignoring the fact that the ranges of values are provided to convey uncertainty rather than equal choice (e.g. Eagle and Thompson Jr 2003). Finally, previous overfishing and overcapacity is a powerful incentive to choose a higher quota (Eagle and Thompson Jr 2003). This practice must stop if countries are to incorporate an ecosystem-based approach to management.

Conclusions

The following recommendations for the management of forage fish are based on the above review.

- **Use a pragmatic approach that includes the most important factors of mortality and change**

The most successful fisheries are managed with conservative exploitation rates to account for possible mistakes in the data or the models used. Successful management schemes are not necessarily the most complicated. Flexibility and simple rules can be very efficient. The key points are the use (and gathering) of ecological and auxiliary data to inform the process, implementing the harvest control rules, and keeping the harvest at a low rate. Rebuilding stocks necessarily entails lower fishing mortality rates.

Ecosystem-based management will not be implemented with all-encompassing models but with management structures that include incrementally the most important factors in the management decision. Nonetheless, there is no ecosystem-based management if we cannot even implement the single-species recommendations.

- **Account for predation in quota setting**

Predators' needs must be taken into account in the management of forage fish because of the impact on other valuable fisheries and of the possible impact on the ecosystem structure. This will likely mean more conservative fishing rates. The incremental fishing rates as a function of the knowledge on the forage species and their ecosystem proposed by Pikitch et al. (2012a) may seem excessive but not in the light of: 1. the difficulty in reducing quotas when fishing capacity and (excessive) quotas have already established socio-economic expectations and demands; 2. the risk to both the ecosystem and the forage fish. It must be argued that it is worth being precautionary to protect both the fishing communities and the ecosystem.

- **Use several sources of information outside fishery data**

When assessments are possible, the most reliable include several abundance indices independent of the fisheries statistics. Assessments with one abundance index are susceptible to failing when distribution or another factor is modified (e.g. NW Atlantic mackerel). When a fishery relies on newly recruited fish, the fishery can be effectively managed on the basis of one or several in season surveys that help define the quota (e.g. herring in Alaska, Humboldt and Benguela upwelling).

- **Acquire ecological knowledge of the ecosystem that allows a broader perspective and explain sudden changes**

Sooner or later, environmental factors, predation or both have to be taken into account in each ecosystem. This became obvious earlier in upwelling areas and extreme environment such as the Antarctic, the Barents Sea, the Baltic, or the Black sea. The most successful cases were those ecosystems where ecological data were gathered to answer the questions posed by collapses.

Spatially-structured information on catch and ecology has been crucial in ecosystem-based management for a variety of reasons. Information on fish distribution and their habitat was instrumental in identifying environmentally driven changes in distribution and decrease in overlap between prey and predators, or identifying sub-stocks. It was also essential to identify possible closed areas that can help protect stock production.

Canada: In spite of numerous policy and strategic documents, Canada is part of the group of countries that do not account for the special situation of forage fish in terms of management. There are no provisions for predators or large uncertainty. Furthermore, only herring, capelin, and mackerel and shrimp are followed and assessed. With the exception of shrimp, the current status of most of these fisheries is uncertain or of concern owing to factors including changes in the environment that are not fully understood. In most cases, the reference points are absent or ad hoc.

All other important forage species such as sand lance, arctic cod and migratory forage species such as shad and alewives, are largely ignored. The authorized fishery on migratory species is based on seasons and effort control.

It is becoming urgent to develop special considerations for forage fishes and fishing. According to the Canadian Policy on new fisheries for forage fish species and to this study, proper management rules require a minimum level of information on the population dynamics and predation of these species. Currently, the lack of knowledge is pervasive. Unfortunately, these policies do not apply to existing fisheries.

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APPENDIX 1. F_{MSY} AND $F_{0.1}$

Two kinds of models have been used in fisheries to estimate F_{MSY} and $F_{0.1}$: biomass dynamic and age-structured. We briefly describe the derivation of F_{MSY} and $F_{0.1}$ using basic models. The biomass dynamic models are based on the concept of stock productivity (growth) and maximal biomass (carrying capacity). As the population size increases, growth decreases as a result of density-dependence and that production is at its highest at a fraction of the biomass at carrying capacity (0.5 using the Schaefer model) – see Figure. A1.1. This highest production point is also the maximum sustainable yield (MSY). It has been noted that the curve is rather flat around MSY which means that it is possible to reduce fishing effort somewhat without losing too much yield and leave more fish in the water.

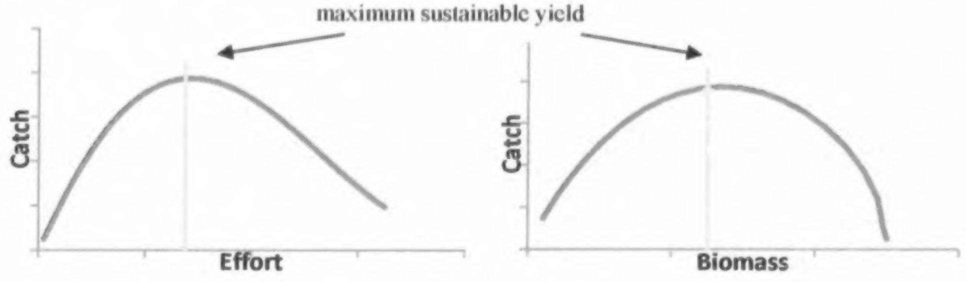

Figure A1.1. Relation between catch and effort and catch and biomass using a surplus production model.

An age structured-model (yield-per-recruit) follows the number of fish of a cohort (individuals born in year x) as they age. Over time, they die from natural (predation, diseases, etc.) and fishing. The rate of decrease will change as a function of mortality (Figure A1.2). Under this model, a population is composed of several cohorts with the same initial number (constant recruitment) and same mortality rate. As the model assumes a constant recruitment in the population, the abundance is relative and expressed as numbers per recruit or biomass per recruit.

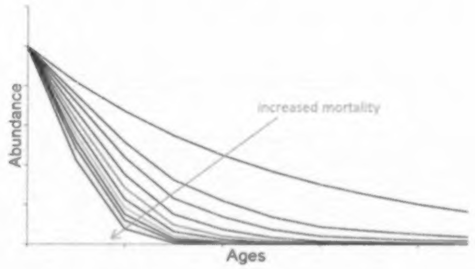

Figure A1.2. Abundance at age for increasing levels of mortality.

Knowing these parameters it is possible to find the maximum yield that a cohort can produce and the corresponding fishing mortality (F_{max}). It was proposed to use a more cautious point called $F_{0.1}$ (Figure A1.3) defined as the point on the curve that corresponds to the value of fishing mortality (F) where the slope is one tenth of the slope at the initial slope.

The well-known F_{MSY} is difficult to determine without knowledge of the relationship of yield and effort (starting from unfished stock, or without times series offering sufficient contrasts to be informative (Hilborn and Walters 1992, chapter 3) to determine unfished biomass and intrinsic rate of growth of the population, thus providing information on the population's level of density dependence (Clark 1991). F_{MSY} is often thought to be too optimistic and thus, increasingly interpreted as a limit rate to be avoided rather than a target (Mace 2001). Thus, several arbitrary rules have been devised to serve as a proxy for F_{MSY} (Clark 1991) or to derive a lower fishing rate that would decrease the risk of stock collapse.

$F_{0.1}$ is used widely in eastern Canada and around the world (Hilborn and Walters 1992) and recently, the European Commission (and ICES) opted to use $F_{0.1}$ as a proxy for F_{MSY} (European Commission 2006). It was set arbitrarily (Gulland and Boerema 1973; Clark 1991) to obtain a more conservative fishing rate. Estimates of F0.1 are sensitive to estimates of natural mortality and selectivity (Deriso 1987; ICES 2010a).

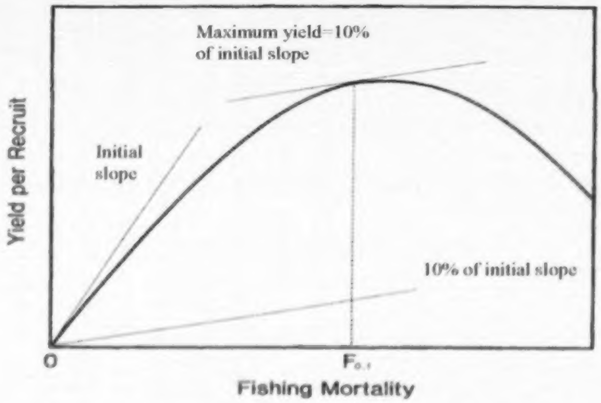

Figure A1.3. The derivation of F0.1 (taken from Deriso 1987).

In an attempt to obtain more robust reference points, scientists used a combination of yield per recruit models with a large array of hypotheses pertaining to the relationship between the numbers of spawners and the recruitment. For instance, Clark (1991) used yield-per-recruit models with typical groundfish life history parameters and a large array of stock-recruitment curves[50]. Results show that $F_{0.1}$ is close to *M*. In addition, $F_{0.1}$ was also close to the "maximum yield fishing rate" that would reduce the spawning biomass per recruit to about 35% ($F_{35\%}$) of the unfished biomass (range 20-60%) and would produce a yield of at least 75% of the maximum yield[51]. This reference point was used to maximize a large fraction of the MSY in cases where F_{MSY} was not available. However, experience showed that this reference point may result in exploitation rates that are too high and in undesirably low biomasses because it is not possible to find a level of fishing that will be safe to use in all case scenarios of stock-recruitment-curves (Clark 2002). Clark's (2002) conclusions regarding the importance of maintaining higher biomass levels and limiting fishing mortality below F_{MSY} have become important objectives in spite of the concurrent loss of yield.

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[50] Stock –recruit relationships are often not available or not informative given short time series, large natural variations or the range of observed stock biomasses is too limited (Clark 1991).

[51] when age at maturity and age of recruitment are similar

APPENDIX 2. CANADIAN GUIDELINES

Provisional Harvest Rule:

In absence of a pre-agreed harvest rule developed in the context of the precautionary approach, a provisional removal reference or fishing mortality (Fp) could be used to guide management and to assess harvest in relation to sustainability. The provisional harvest rule is as follows depending on the stock status:

In the "Healthy Zone": $Fp < F_{MSY}$
In the "Cautious Zone": $Fp < F_{MSY} \times ((\text{Biomass} - 40\% B_{MSY}) / (80\% B_{MSY} - 40\% B_{MSY}))$
In the "Critical Zone": $Fp = 0$

In absence of estimates related to the status of the stock and of the fishery at MSY, options for provisional estimates of B_{MSY} and F_{MSY} are provided below.

Biomass at MSY.

In absence of an estimate of B_{MSY} from an explicit model, the provisional estimate of B_{MSY} could be taken as follows (select the first feasible option):

- The biomass corresponding to the biomass per recruit at $F_{0.1}$ multiplied by the average number of recruits; or
- The average biomass (or index of biomass) over a productive period; or
- The biomass corresponding to 50% of the maximum historical biomass.

Fishing mortality at MSY.

In absence of an estimate of F_{MSY} from an explicit model, the provisional estimate of F_{MSY} could be taken as follows (select the first feasible option):

- The fishing mortality corresponding to $F_{0.1}$; or
- The average fishing mortality (or an index of fishing mortality) that did not lead to stock decline over a productive period; or
- The fishing mortality equal to natural mortality inferred from life history characteristics of the species.

APPENDIX 3. POTENTIAL REMOVAL AND PREDATORS

Box 1: The Potential Biological Removal (PBR) (Wade 1998)

This method was developed to assess when the incidental deaths of marine mammals were too excessive to maintain or promote the recovery of a population, in absence of good population estimates and of long time series. The *PBR* is maximum number of animals that can be removed from a population not including natural mortalities. It is a function of the minimum estimate of population size *Nmin*, the maximum rate of population increase *Rmax*, and mitigated with a recovery factor *Fr*

$$PBR = Nmin * 0.5 Rmax * Fr$$

where *Fr* is between 0 and 1, and *Rmax* is set at 0.04 for cetaceans and 0.12 for pinnipeds. The author ran multiple simulations using a logistic model with initial populations of 30% of carrying capacity to examine the effects of various sources of uncertainty and biases on the ability of the population to recover to the point of maximum production. The study concluded that *Nmin* should be estimated as the 20th percentile of the confidence limit of the population estimate, and that Fr =0.5 allows the management goals to be reached despite various sources of bias.

Box 2. Fowler's idea (Fowler and McCluskey 2011)

Fowler and McCluskey (2011) start from the principle that humans have an abnormal influence on ecosystems and that present data and ecosystem status reflect these past influences. They suggest choosing a harvest rate that would result in a yield no greater than the mean annual consumption of the prey by individual marine predator. The mean could be defined as the geometric mean or the 95% confidence limit of the arithmetic mean, for instance.

In the case of the Gulf of Maine herring, for example, the authors show that between 1988 and 1991, fishing removes 64 times more herring than the geometric mean of consumption (removals) by the 12 predators (marine mammals, birds, fish) and 8.8 times higher than the largest consumption recorded for a non-human predator. Thus, Fowler and McCluskey recommend that fisheries catch should be reduced by over 98% to correspond to the geometric mean and increase the "diversity index" of herring predators.
